

AD-A182 298

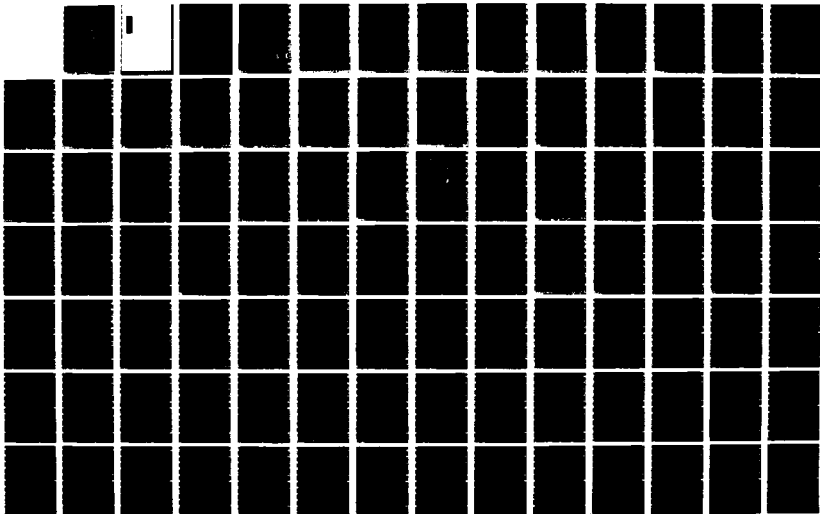
ENVIRONMENTAL AND WATER QUALITY OPERATIONAL STUDIES:
LIMNOLOGICAL STUDIES (U) ARMY ENGINEER WATERWAYS
EXPERIMENT STATION VICKSBURG MS ENVIR

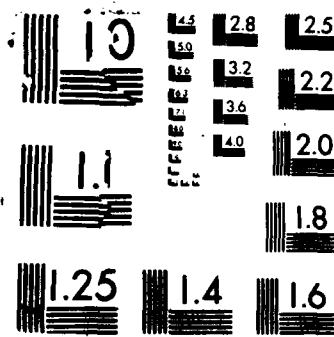
1/2

UNCLASSIFIED

R H KENNEDY ET AL MAR 87 WES/TR/E-85-2-2 F/G 8/8

NL





Results and Discussion

15. Hydrographs for each of the tributary streams were characterized by peaks following spring rains and snowmelt, and low, relatively stable flows during other seasons (Figure 2). Flows for the Eau Galle River were highest, averaging 10 to 20 times those for the other tributaries.

16. Patterns of change in the concentrations of total phosphorus, total nitrogen, and total organic carbon with changes in flow were highly variable (Figures 3-6). In general, total phosphorus increased with increased flow; however, variability in concentration was greatest

Destroy this report when no longer needed. Do not
return it to the originator.

The findings in this report are not to be construed as an
official Department of the Army position unless so
designated by other authorized documents.

The contents of this report are not to be used for
advertising, publication, or promotional purposes.
Citation of trade names does not constitute an
official endorsement or approval of the use of such
commercial products.

12

Unclassified

SECURITY CLASSIFICATION OF THIS PAGE

REPORT DOCUMENTATION PAGE				Form Approved OMB No. 0704-0188 Exp Date Jun 30, 1986	
1a REPORT SECURITY CLASSIFICATION Unclassified			1b RESTRICTIVE MARKINGS		
2a SECURITY CLASSIFICATION AUTHORITY			3 DISTRIBUTION/AVAILABILITY OF REPORT Approved for public release; distribution unlimited.		
2b DECLASSIFICATION/DOWNGRADING SCHEDULE			5 MONITORING ORGANIZATION REPORT NUMBER(S)		
4 PERFORMING ORGANIZATION REPORT NUMBER(S) Technical Report E-85-2			5 MONITORING ORGANIZATION REPORT NUMBER(S)		
6a NAME OF PERFORMING ORGANIZATION USAEWES Environmental Laboratory		6b OFFICE SYMBOL (If applicable)	7a NAME OF MONITORING ORGANIZATION		
6c ADDRESS (City, State, and ZIP Code) PO Box 631 Vicksburg, MS 39180-0631			7b ADDRESS (City, State, and ZIP Code)		
8a NAME OF FUNDING/SPONSORING ORGANIZATION US Army Corps of Engineers		8b OFFICE SYMBOL (If applicable)	9 PROCUREMENT INSTRUMENT IDENTIFICATION NUMBER		
8c ADDRESS (City, State, and ZIP Code) Washington, DC 20314-1000			10 SOURCE OF FUNDING NUMBERS		WORK UNIT ACCESSION NO EWQOS VIIA
11 TITLE (Include Security Classification) Limnological Studies at Eau Galle Lake, Wisconsin; Report 2, Special Studies and Summary					
12 PERSONAL AUTHOR(S) Kennedy, Robert H., and Gunkel, Robert C., Jr.					
13a TYPE OF REPORT Report 2 of a series		13b TIME COVERED FROM _____ TO _____	14 DATE OF REPORT (Year, Month, Day) March 1987		15 PAGE COUNT 171
16 SUPPLEMENTARY NOTATION Available from National Technical Information Service, 5285 Port Royal Road, Springfield, VA 22161.					
17 COSATI CODES			18 SUBJECT TERMS (Continue on reverse if necessary and identify by block number)		
FIELD	GROUP	SUB-GROUP	Eau Galle Lake Loading Macroinvertebrates Macrophytes Mixing events (Continued) Phosphorus		
19 ABSTRACT (Continue on reverse if necessary and identify by block number) Eau Galle Lake was one of four Corps of Engineers reservoirs surveyed during the Environmental and Water Quality Operational Studies. Eau Galle Lake is a small flood control reservoir that receives high nutrient loads, experiences hypolimnetic anoxia, and has abundant macrophyte and algal production. This report documents the studies of biological communities and physicochemical and biological processes influencing water quality conditions in Eau Galle Lake.					
20 DISTRIBUTION/AVAILABILITY OF ABSTRACT <input checked="" type="checkbox"/> UNCLASSIFIED/UNLIMITED <input type="checkbox"/> SAME AS RPT <input type="checkbox"/> DTIC USERS			21 ABSTRACT SECURITY CLASSIFICATION Unclassified		
22a NAME OF RESPONSIBLE INDIVIDUAL			22b TELEPHONE (Include Area Code)		22c OFFICE SYMBOL

DD FORM 1473, 84 MAR

83 APR edition may be used until exhausted
All other editions are obsoleteSECURITY CLASSIFICATION OF THIS PAGE
Unclassified

DATE
JUL 15 1987

Unclassified

SECURITY CLASSIFICATION OF THIS PAGE

18. SUBJECT TERMS (Continued).

Phytoplankton	Sediment distribution
Reservoir	Sediment quality
Sediment deposition	Zooplankton

Unclassified

SECURITY CLASSIFICATION OF THIS PAGE

PREFACE

The work described in this report is part of the Environmental and Water Quality Operational Studies (EWQOS) Program, Work Unit VIIA, Reservoir Field Studies, conducted by the US Army Engineer Waterways Experiment Station (WES) for the Office, Chief of Engineers (OCE), US Army. The OCE Technical Monitors were Dr. John Bushman, Mr. Earl Eiker, and Mr. James L. Gottesman.

The study was completed by the Aquatic Processes and Effects Group (APEG), Environmental Research and Simulation Division (ERSD), Environmental Laboratory (EL), WES. The report was prepared by Dr. Robert H. Kennedy, Mr. Robert C. Gunkel, Jr., Mr. Joe H. Carroll, Dr. John W. Barko, Mr. Derrick J. Bates, Dr. Gerald J. Filbin, Ms. Susan M. Hennington, Ms. Dwilette G. McFarland, Mr. William F. James, and Dr. Robert F. Gaugush, APEG; Dr. John Nestler, Water Quality Modeling Group, ERSD; and Dr. Paul J. Garrison, Wisconsin Department of Natural Resources. Dr. Kennedy and Mr. Gunkel were editors. The report was prepared under the supervision of Dr. Thomas Hart, Chief, APEG; Mr. Donald L. Robey, Chief, ERSD; and Dr. John Harrison, Chief, EL. Dr. Jerome L. Mahloch was Program Manager of EWQOS. The manuscript was edited by Ms. Jessica S. Ruff, Information Products Division, WES.

COL Allen F. Grum, USA, was the previous Director of WES. COL Dwayne G. Lee, CE, is the present Commander and Director. Dr. Robert W. Whalin is Technical Director.

This report should be cited as follows:

Kennedy, R. H., and Gunkel, R. C., Jr., eds. 1987. "Limnological Studies at Eau Galle Lake, Wisconsin; Report 2, Special Studies and Summary," Technical Report E-85-2, US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

CONTENTS

	<u>Page</u>
PREFACE.	1
PART I: INTRODUCTION	4
Background	4
Scope	7
References	8
PART II: MATERIAL LOADINGS TO EAU GALLE LAKE	9
Introduction	9
Methods	10
Results and Discussion	11
Conclusions	21
References	22
PART III: SEASONAL GROWTH AND COMMUNITY COMPOSITION OF PHYTOPLANKTON	24
Introduction	24
Methods and Materials	24
Results	26
Discussion	33
References	40
PART IV: THE ZOOPLANKTON COMMUNITY OF EAU GALLE LAKE	44
Introduction	44
Methods	44
Results	45
Discussion	56
Summary	61
References	62
PART V: GROWTH AND NUTRITION OF SUBMERSED AQUATIC MACROPHYTES	64
Introduction	64
Materials and Methods	65
Results	66
Discussion	69
Summary and Conclusions	72
References	72
PART VI: FACTORS DETERMINING THE DISTRIBUTION OF MACROINVERTE- BRATES ASSOCIATED WITH MACROPHYTES OF THE LITTORAL ZONE IN EAU GALLE LAKE	76
Introduction	76
Materials and Methods	78
Results	81
Discussion	84
Conclusions	90

	<u>Page</u>
References	90
PART VII: SEASONAL PATTERNS OF SEDIMENT DEPOSITION IN EAU GALLE LAKE	92
Introduction	92
Methods	93
Results	95
Discussion	108
Conclusions	112
References	112
PART VIII: SEDIMENT DISTRIBUTION AND QUALITY	115
Introduction	115
Methods and Materials	116
Results and Discussion	118
Conclusions	125
References	126
PART IX: HYPOLIMNETIC PHOSPHORUS DYNAMICS	128
Introduction	128
Methods	129
Results and Discussion	131
Summary	142
References	147
PART X: MIXING EVENTS IN EAU GALLE LAKE	149
Introduction	149
Methods	150
Results and Discussion	152
Conclusions.	162
References	163
PART XI: SUMMARY	164



Accession For	
NTIS GRA&I	<input checked="" type="checkbox"/>
DTIC TAB	<input type="checkbox"/>
Unannounced	<input type="checkbox"/>
Justification	
By	
Distribution/	
Availability Codes	
Dist	Avail and/or Special
A-1	

LIMNOLOGICAL STUDIES AT EAU GALLE LAKE, WISCONSIN

SPECIAL STUDIES AND SUMMARY

PART I: INTRODUCTION*

Background

1. The influx of nutrients, organic material, and sediment to lakes and reservoirs has led to a progressive deterioration of these valuable water resources. Increased nutrient levels promote increased algal production, resulting in reduced water transparency, taste and odor problems, diminished aesthetic value, and reduced oxygen concentrations in bottom waters. These problems are further aggravated by excessive inputs of sediments and oxygen-demanding organic material. While increased public awareness of these problems has prompted the establishment of new research programs and the passage of water resource legislation, the need for improved management strategies for maintaining or enhancing the quality of these resources still exists.

2. The US Army Corps of Engineers (CE) presently maintains and operates over 400 reservoir projects for the purposes of flood control, hydropower, navigation, water supply, and recreation. In an effort to better define reservoir water quality problems and to develop technological and managerial alternatives for ameliorating these problems in a manner consistent with project purposes, the CE initiated the Environmental and Water Quality Operational Studies (EWQOS) Program (Keeley et al. 1978). A major portion of this program consisted of limnological studies at four characteristic CE reservoirs. These included Lake Red-rock, a flood control impoundment of the Des Moines River in central Iowa; DeGray Lake, a large hydropower project located in a forested watershed in south-central Arkansas; West Point Lake, a large hydropower

* Part I was written by Robert H. Kennedy, Robert C. Gunkel, Jr., and Joseph H. Carroll.

project created by impoundment of the Chattahoochee River approximately 100 km downstream from Atlanta, Ga.; and Eau Galle Lake, a small, eutrophic flood control reservoir in west-central Wisconsin.

3. Studies conducted at each of these sites since 1978 have been of two basic types: long-term monitoring studies and short-term, process-oriented studies. Long-term monitoring studies provided data needed for the calibration and evaluation of several ecological models as well as information concerning general water quality characteristics and trends. Short-term studies, often conducted during specific seasons of the year or under a predetermined set of hydrometeorologic conditions, were aimed at defining processes impacting reservoir water quality. These types of studies provided a means for improving existing water quality models and have greatly increased the understanding of reservoir ecosystem processes.

4. Since Eau Galle Lake was the smallest site investigated under the EWQOS Program and therefore posed the least logistical and sampling problems, many of the intensive, process-oriented studies were conducted here. In order to accomplish this, a field analytical laboratory was established at the site, as well as facilities and equipment for obtaining meteorologic and hydrologic data. The laboratory facility, which consisted of a completely equipped analytical laboratory and associated office and storage facilities, provided necessary capabilities for a wide variety of water quality analyses. Since laboratory and field personnel were located at the site, a wide range of field activities were possible.

5. Limnological and hydrological data were collected routinely in Eau Galle Lake, all major tributaries, and the lake's discharge. The locations of major sampling stations discussed throughout this report are shown in Figure 1. Routine sampling was initiated in November 1980 and continued through October 1982. Sampling involved the measurement of in situ variables (i.e., dissolved oxygen, temperature, pH, and specific conductance) at weekly intervals and the collection of water samples for chemical analyses at 2-week intervals. Chemical analyses included nutrients, metals, solids, and biological pigments. A complete

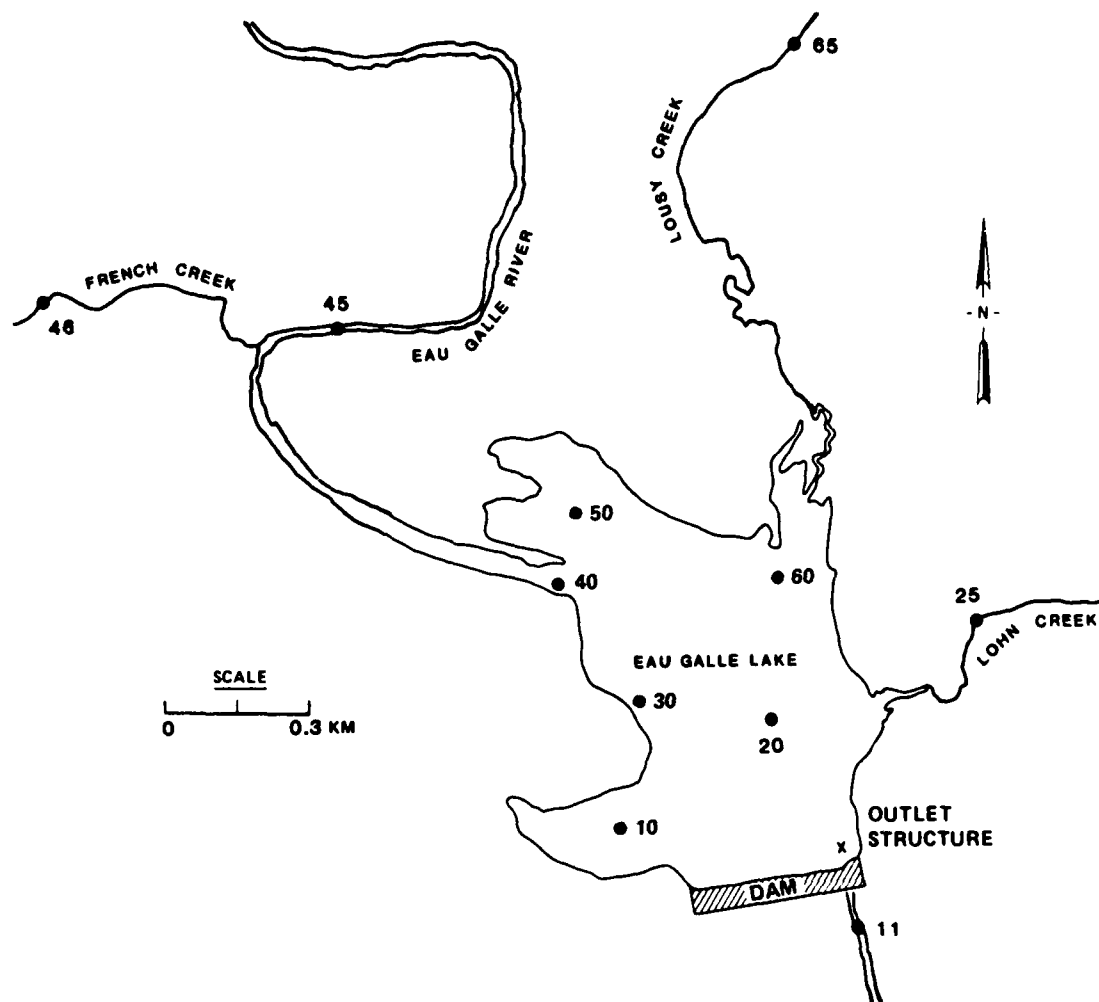


Figure 1. Eau Galle Lake sampling stations

discussion of field and laboratory methods and a listing of analyses performed are presented in the first report in this series (Kennedy 1985).

6. Several general conclusions were drawn from information obtained as a result of routine monitoring efforts (Kennedy 1985):

- a. Morphometric characteristics of the lake basin reflect excavations and channel diversions during construction. These characteristics include a deep central basin, a deep narrow approach channel, and modifications to the river channel near the river inflow. Preliminary findings indicate that these characteristics and the operation of the outlet structure strongly influence in-lake flow patterns.

- b. Seasonal variability in flow and material concentration reflects geologic and land use patterns in each tributary basin and the impacts of spring snowmelt events. In general, material loadings were highest during late winter and early spring, coincident with snowmelt and spring rains. Within these high-flow events, changes in concentrations are predictable; dissolved constituent concentrations are highest prior to peaks in the hydrograph, while particulate concentrations are highest coincident with peaks in flow.
- c. Seasonal variations in flow and material loads had pronounced impacts on lake conditions; most notable were impacts of snowmelt and runoff. During these periods, inflow entered the lake as an overflowing density current or mixed completely with lake waters. In addition to increases in lake level and discharges, these events markedly increased lake nutrient, metal, and solids concentrations. Increases were also apparent in the discharge. Based on comparisons with other loading events, these events supply a significant portion of the total annual load and thus greatly influence lake nutrient levels prior to the summer growing season.
- d. Few differences were observed between lake sampling stations, suggesting that conditions observed at station 20 were indicative of conditions in the lake as a whole. Differences observed for stations 40 and 50 reflect influences of the inflowing river and the growth of macrophytes in the shallow northern cove, respectively. Marked vertical differences were observed seasonally. Thermal stratification, although relatively weak, fostered the establishment of an anoxic hypolimnion and subsequent releases from sediments of nutrients and metals.

Scope

7. Reported here are the results of special studies designed to provide detailed information concerning selected processes influencing water quality conditions in Eau Galle Lake. While each of these studies was conducted during the same period, specific sampling schedules and locations, as well as methods, often differed from those employed during the routine monitoring studies.

8. Emphasis in this report is placed on descriptions of biological communities (zooplankton, phytoplankton, and macrophytes) and processes influencing the fate and transport of materials (material

loading, sedimentation, and mixing). Together with previously reported results of the routine monitoring studies, these data allow for the detailed description of water quality conditions in Eau Galle Lake.

References

Keeley, J. W., J. L. Mahloch, J. W. Barko, D. Gunnison, and J. D. Westhoff. 1978. Identification and assessment of environmental quality problems and research development. Technical Report E-78-1. US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Kennedy, R. H., ed. 1985. Limnological studies at Eau Galle Lake, Wisconsin; Report 1, Introduction and water quality monitoring studies. Technical Report E-85-2. US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

PART II: MATERIAL LOADINGS TO EAU GALLE LAKE*

Introduction

9. Lakes and reservoirs occupy depressions in the landscape and, therefore, are the recipients of water and materials exported from the surrounding watershed by streams and rivers. It is through this linkage that the nutrient status and productivity of lakes and reservoirs are influenced by natural and man-induced processes in the watershed (Likens and Bormann 1974). Problem conditions often occur in the receiving lake or reservoir when exports of sediment and growth-stimulating plant nutrients, particularly phosphorus, are excessive. These problems include decreased volume, excessive growth of algae and macrophytes, reduced water clarity, and reduced dissolved oxygen concentrations in bottom waters during periods of stratification.

10. The linkage between terrestrial exports and the trophic characteristics of the receiving water body forms the basis for methods commonly used for evaluating and/or predicting lake and reservoir problems. In general, these methods involve quantification of annual water and material loads as a means of evaluating or estimating actual or potential material concentrations in the lake or reservoir as they may relate to such water quality conditions as chlorophyll standing crop, transparency, and/or oxygen depletion (e.g., Walker 1985, 1986; Reckhow and Chapra 1983).

11. Reported here are results of efforts to quantify total phosphorus, total nitrogen, and total organic carbon loads to Eau Galle Lake. Estimates are for the period 1981 to 1982, a period when comprehensive limnological studies were conducted at Eau Galle Lake.

* Part II was written by Robert H. Kennedy.

Methods

12. Daily streamflows for each of the major tributaries to Eau Galle Lake were determined based on stage data recorded continuously at stream gages installed and operated by the US Geologic Survey (see Figure 1 for gaging locations). Gage data for French Creek (station 46), Lousy Creek (station 65), and Lohn Creek (station 25) were available for the period January 1981 to December 1982. Gage data for the Eau Galle River immediately upstream from the lake (station 45) were available only for the period November 1981 to December 1982; flows for the period January 1981 to October 1981 were calculated by linear regression analysis and were based on flows observed near Woodville, Wis. Hourly flows recorded at each gage were averaged and recorded as average daily flow.

13. Water quality samples were collected at each station at approximately 2-week intervals. Approximately 45 samples were collected at stations 25, 46, and 65 during the period November 1980 to September 1982; 25 samples were collected at station 45 between November 1981 and October 1982. Additional information was obtained as a result of intensive (4 to 12 samples per day) water quality surveys conducted at stations 45 and 65 during a snowmelt and runoff event which occurred during late March and early May 1982, respectively. Sample collection and analyses for total phosphorus, total nitrogen, and total organic carbon followed standard methods as described in Johnson and Lauer (1985).

14. Material fluxes (kilograms per year) were computed by one of five methods using the FLUX data reduction program (Walker 1986). These methods include direct and flow-weighted averages, a modified ratio estimate, and first- and second-order regression analysis. Briefly, fluxes were computed by first determining the best method for describing fluxes for paired observations of instantaneous flow and concentration. The relative merits of each method were based on measures of variance and comparisons of observed and predicted fluxes. Annual fluxes were then computed by applying the chosen relationship to the continuously recorded flow data.

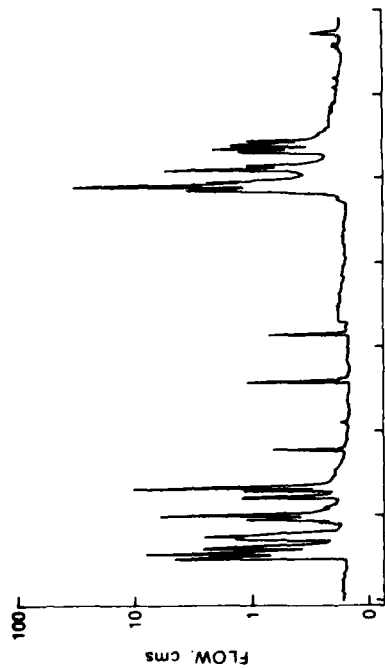
Results and Discussion

15. Hydrographs for each of the tributary streams were characterized by peaks following spring rains and snowmelt, and low, relatively stable flows during other seasons (Figure 2). Flows for the Eau Galle River were highest, averaging 10 to 20 times those for the other tributaries.

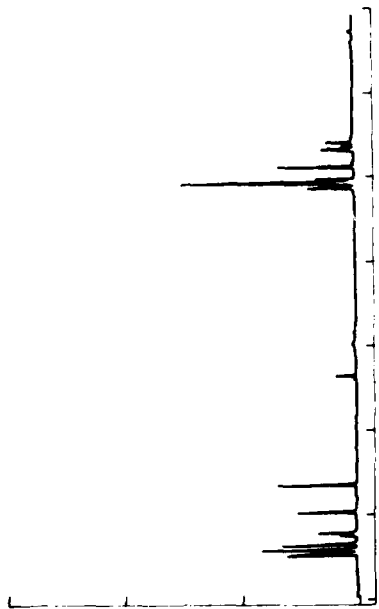
16. Patterns of change in the concentrations of total phosphorus, total nitrogen, and total organic carbon with changes in flow were highly variable (Figures 3-6). In general, total phosphorus increased with increased flow; however, variability in concentration was greatest at higher flows. With the exception of those for the Eau Galle River (station 45), changes in the concentrations of total nitrogen and total organic carbon displayed little relation to changes in flow.

17. Variability in the concentrations of total phosphorus, total nitrogen, and total organic carbon not attributable to changes in flow may have been related, in part, to seasonal influences and watershed land use activities. Likens et al. (1977) document the potential importance of seasonal changes in biological activity in forested watersheds in determining the quantity and form of materials exported during an annual cycle. In northern forested areas, snowmelt runoff also exerts a strong seasonal influence on export rates and quality (Timmons et al. 1977). The influence of snowmelt runoff on the water quality of tributaries to Eau Galle Lake was clearly documented by Ashby and James (1985).

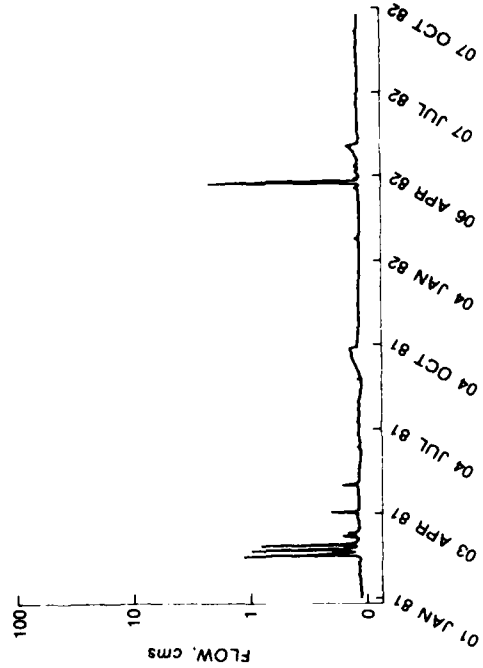
18. Land use patterns in the watershed would also potentially influence the seasonal distribution of nutrient concentrations in tributary streams. Burwell, Timmons, and Holt (1975) and Alberts, Schuman, and Burwell (1978) discuss the influences of cultivation practices on the magnitude and seasonality of nitrogen and phosphorus losses from agricultural watersheds. Losses of organic carbon from cultivated watersheds have also been reported to exhibit seasonal changes related to cover type and cultivation method (Schreiber and McGregor 1979).



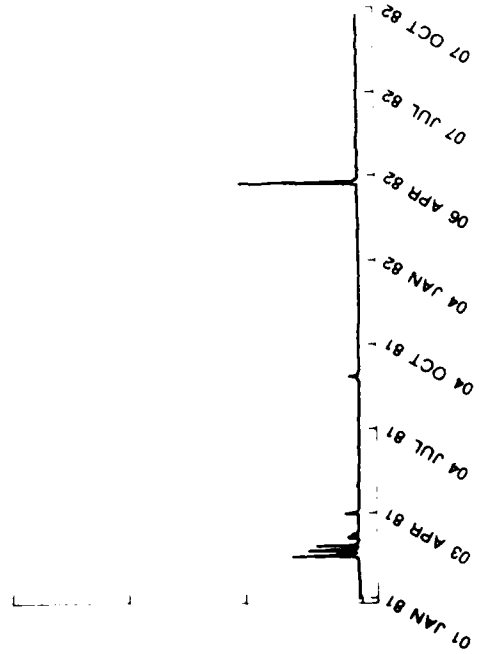
a. EAU GALLE RIVER



b. FRENCH CREEK



c. LOUSY CREEK



d. LOHN CREEK

Figure 2. Daily fluctuations in flow (cubic meters per second) for tributary streams for the period 1981-1982

EAU GALLE RIVER - STATION 45

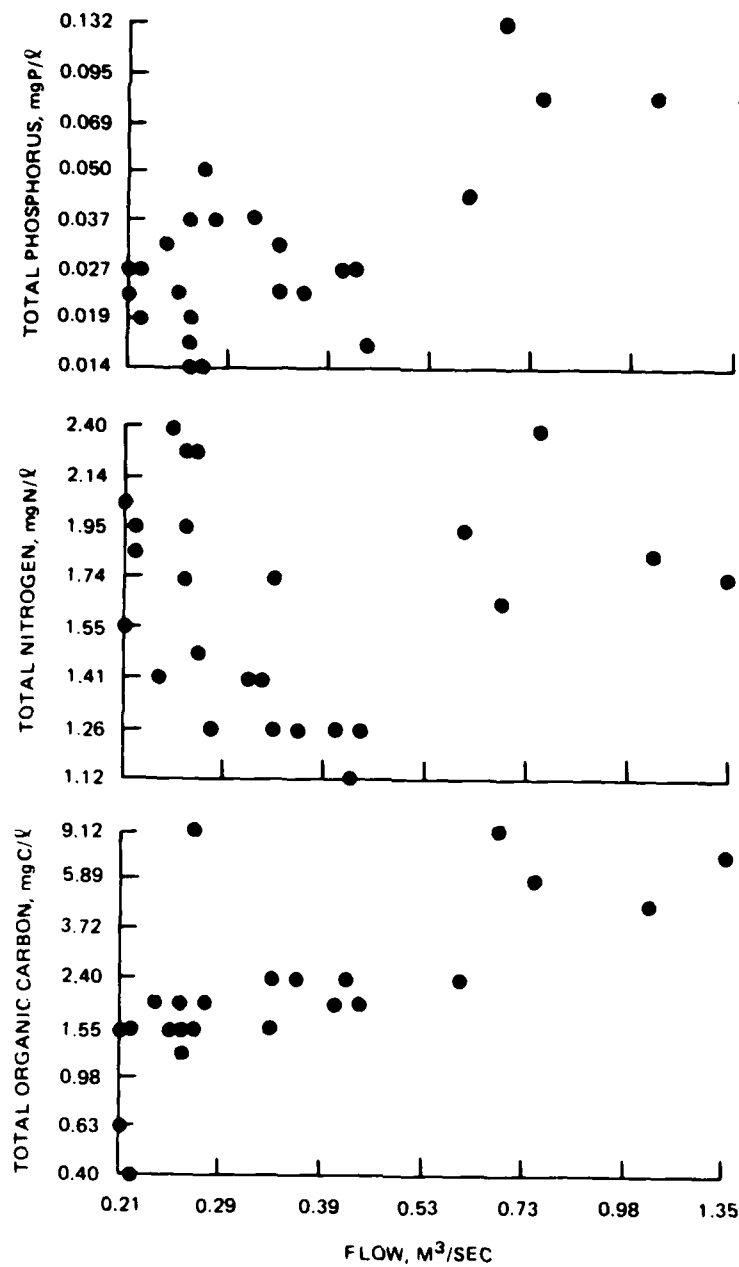


Figure 3. Relations between total phosphorus, nitrogen, and total organic carbon concentrations, and flow for the Eau Galle River (station 45)

FRENCH CREEK - STATION 46

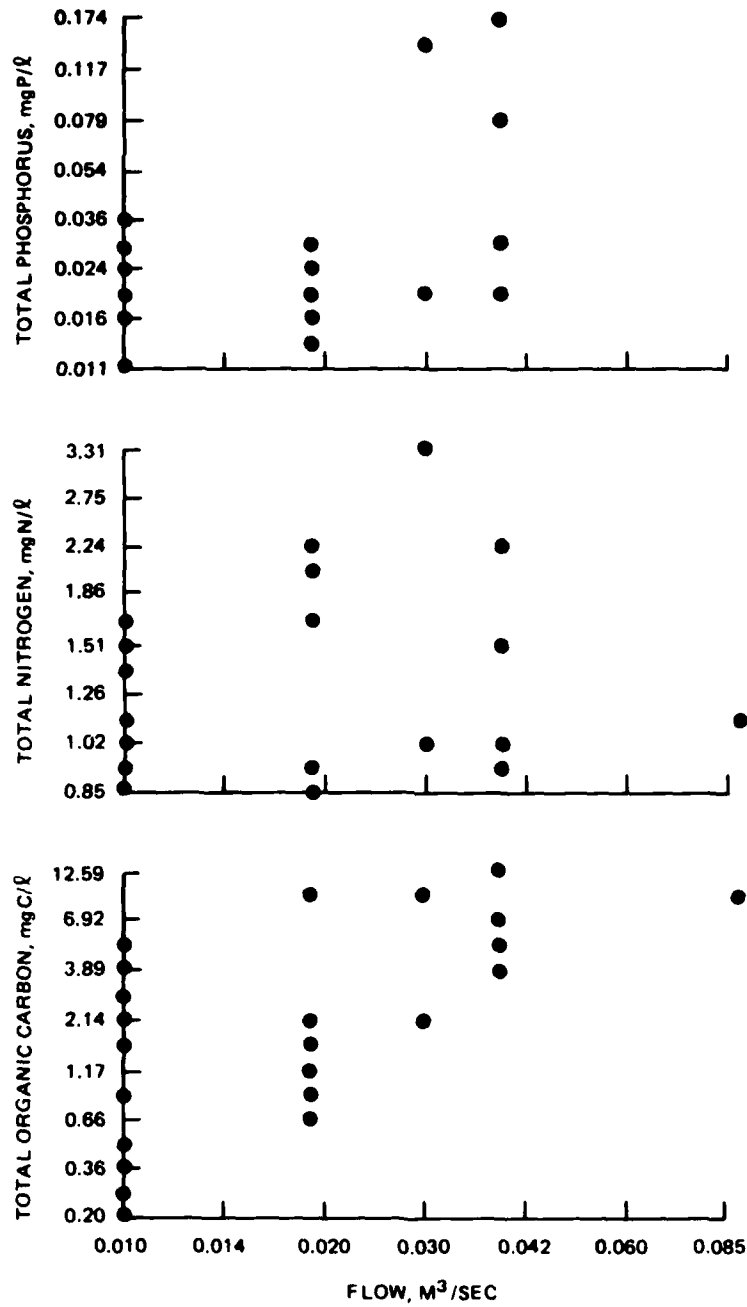


Figure 4. Relations between total phosphorus, total nitrogen, and total organic carbon concentrations, and flow for French Creek (station 46)

LOUSY CREEK - STATION 65

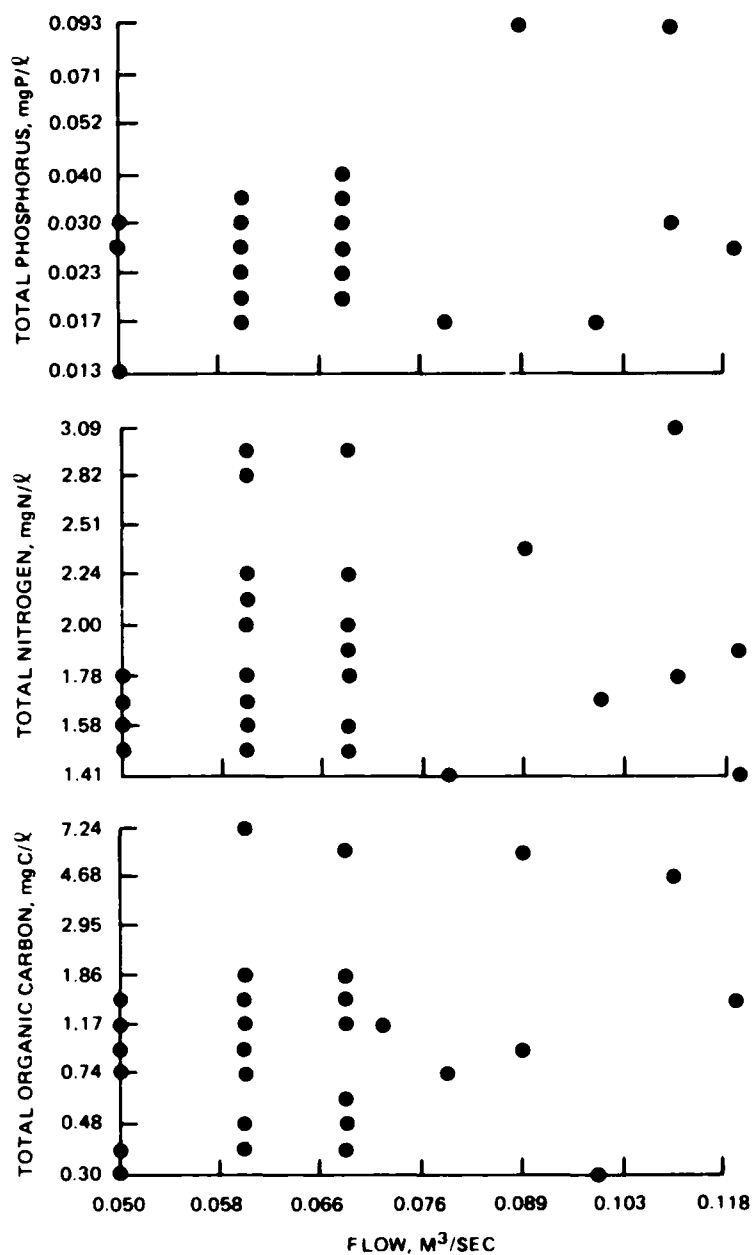


Figure 5. Relations between total phosphorus, total nitrogen, and total organic carbon concentrations, and flow for Lousy Creek (station 65)

LOHN CREEK - STATION 25

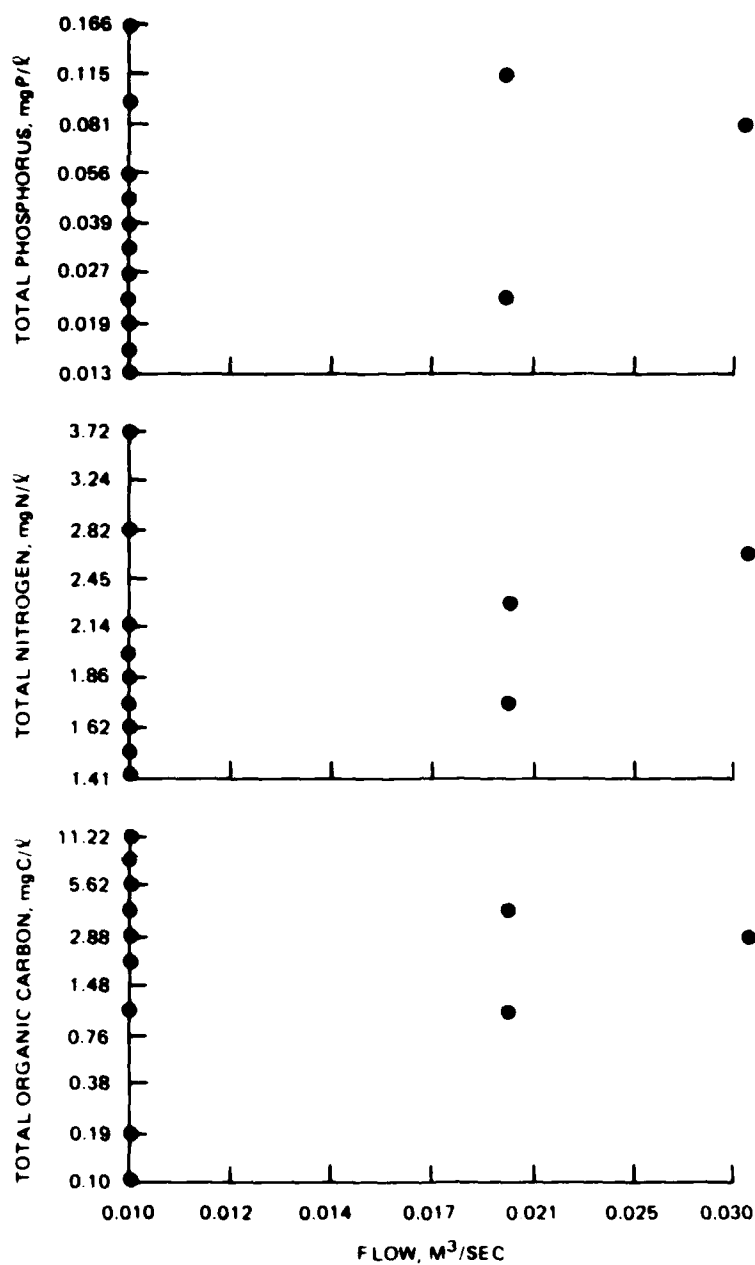


Figure 6. Relations between total phosphorus, total nitrogen, and total organic carbon concentrations, and flow for Lohn Creek (station 25)

19. Total phosphorus, total nitrogen, and total organic carbon loading estimates (kilograms per year) for each of the four tributaries for 1981 and 1982 are presented in Table 1. Also presented are measures of variation (coefficient of variation, CV) for each estimate for each constituent and year, and the regression coefficient (r^2) resulting from comparisons of observed and predicted loads. Omitted from consideration in estimating loads for 1982 were water quality and flow data for the spring snowmelt period (25 March-4 April 1982). Loads during this period were estimated independently as discussed below.

20. In general, loads for the Eau Galle River (station 45) were markedly higher than those for the other tributaries combined. Estimates for this tributary also exhibited the highest degree of agreement between predicted and observed values. Weakest agreement between predicted and observed values was for total phosphorus and total organic carbon for Lousy and Lohn Creeks. For all other estimates, the estimation method chosen accounted for more than 50 percent of the variability in loads calculated from observed pairs of flow and concentration; over 80 percent was accounted for in total phosphorus, total nitrogen, and total organic carbon estimates for the Eau Galle River.

21. Independent estimates of total phosphorus, total nitrogen, and total organic carbon loads for the Eau Galle River and Lousy Creek during the 1982 snowmelt event were obtained as the sum of the products of average daily flow and flow-averaged daily concentrations observed during intensive sampling. These estimates, as well as estimates for which data for the snowmelt event were not considered, are presented in Table 2. Comparisons of these estimates clearly show the importance of this seasonal event to the total annual loading estimate. While the water load for the Eau Galle River increased by a factor of 1.84 when the snowmelt data were considered, estimates of total phosphorus, total nitrogen, and total organic carbon loads increased by factors of 9.58, 2.45, and 4.84, respectively. Increases in the estimates for Lousy Creek, although less pronounced than those for the Eau Galle River, exhibited similar patterns; increases were greatest for total phosphorus and least for total nitrogen.

Table 1
Material Loading Estimates for
Tributaries to Eau Galle Lake

Loading Variable	1981		1982*		r ^{2**}
	Load kg/yr	CV	Load kg/yr	CV	
<u>Eau Galle River (Station 45)</u>					
Flow†	15.32	--	11.54	--	--
Total phosphorus	867.	0.083	551.	0.088	0.89
Total nitrogen	26,430.	0.040	20,360.	0.033	0.92
Total organic carbon	64,563.	0.074	40,867.	0.078	0.80
<u>French Creek (Station 46)</u>					
Flow	0.90	--	1.50	--	--
Total phosphorus	52.	0.158	89.	0.164	0.76
Total nitrogen	1,253.	0.071	2,002.	0.075	0.81
Total organic carbon	5,536.	0.119	9,706.	0.122	0.68
<u>Lousy Creek (Station 65)</u>					
Flow	2.59	--	1.82	--	--
Total phosphorus	82.	0.111	74.	0.105	0.46
Total nitrogen	4,880.	0.047	4,657.	0.041	0.56
Total organic carbon	4,964.	0.121	4,471.	0.127	0.35
<u>Lohn Creek (Station 25)</u>					
Flow	0.60	--	0.50	--	--
Total phosphorus	23.	0.141	20.	0.141	0.42
Total nitrogen	1,116.	0.042	990.	0.042	0.63
Total organic carbon	1,255.	0.154	1,111.	0.154	0.13

* March storm event not included in 1982 water and material loading estimates for Eau Galle River (station 45) or Lousy Creek (station 65).

** Regression coefficient for estimated load versus observed load.

† Water loads have units of 10⁶ m³/year.

Table 2
Relative Importance of the March 1982 Storm Event to the Estimation
of Annual Material Loads for the Eau Galle River (Station 45) and
Lousy Creek (Station 65)

<u>Loading Variable</u>	<u>Loading, kg</u>			<u>Proportional Change in Estimate</u>
	<u>Storm Excluded</u>	<u>Storm Only</u>	<u>Combined Total</u>	
<u>Eau Galle River (Station 45)</u>				
Flow*	11.54	9.64	21.18	1.84
Total phosphorus	551.	4,729.	5,280.	9.58
Total nitrogen	20,360.	29,481.	49,841.	2.45
Total organic carbon	40,867.	156,976.	197,843.	4.84
<u>Lousy Creek (Station 65)</u>				
Flow	1.82	0.57	2.39	1.31
Total phosphorus	74.	207.	281.	3.80
Total nitrogen	4,657.	926.	5583.	1.20
Total organic carbon	4,471.	7,134.	11,605.	2.60

* Water loads have units of 10^6 m^3 .

22. Loading estimates for 1982 for the four tributaries were modified to reflect increases due to the snowmelt event. Estimates for the Eau Galle River and Lousy Creek were increased based on observed data. Those for French Creek and Lohn Creek were increased based on assumed similarities (i.e., subwatershed size and land use patterns) between these tributaries and Lousy Creek. This was accomplished by applying factors of proportional change observed for Lousy Creek (Table 2) to loading estimates for French Creek and Lohn Creek (Table 1).

23. Annual estimates of mass and areal loading for Eau Galle Lake for 1981 and 1982 are presented in Table 3. Water loads for these two years were similar; however, marked differences were apparent for constituent loads, particularly total phosphorus and total organic carbon. Since these differences were primarily attributable to the occurrence of

Table 3
Annual Loading Estimates for Eau Galle Lake
for 1981 and 1982

Constituent	Mass Load kg/yr		Areal Load mg/m ² /yr	
	1981	1982	1981	1982
Water*	19.41	25.57	3.24	4.26
Total phosphorus	1,024.	5,975.	1.71	10.00
Total nitrogen	33,679.	59,014.	56.13	98.36
Total organic carbon	76,318.	237,573.	127.20	395.96

* Mass water load is presented as $10^6 \text{ m}^3/\text{year}$; areal water load was calculated as mass water load divided by lake surface area and is presented as meters per year.

a rapid snowmelt event, it is clear that spring events, which result in rapid runoff from land surfaces, play a dominant role in determining loadings to this lake. Areal water loads (32.4 and 42.6 m/year for 1981 and 1982, respectively) and total phosphorus areal loads (1.71 and 9.96 gm/m²/year for 1981 and 1982, respectively) are consistent with the observed eutrophic status of this lake (cf. Vollenweider 1975).

24. While watershed export coefficients for each constituent were highly variable between years (Table 4), estimates were similar among subwatersheds and were within the range of values reported for other watersheds having similar land use patterns. Phosphorus export coefficients, which ranged from 0.03 to 0.40 kg P/ha/year, and nitrogen export coefficients, which ranged from 1.14 to 3.80 kg N/ha/year, were comparable to values reported by Beaulac and Reckhow (1982) for forested and pasture land. Both of these land uses predominate in the Eau Galle watershed. Total organic carbon export coefficients ranged from 1.79 to 22.94 kg C/ha/yr and were, on the average, below the values reported by Schreiber and McGregor (1979) for agricultural lands. This observation is also consistent with land use patterns in this watershed.

Table 4
Land Use and Nutrient Export Relationships
for the Eau Galle Lake Watershed

<u>Subwatershed</u>	<u>Area, ha</u>	<u>Export Coefficient, kg/ha/yr*</u>		
		<u>Phosphorus</u>	<u>Nitrogen</u>	<u>Carbon</u>
Eau Galle River	13,100	0.07,0.40	2.02,3.80	4.93,15.10
French Creek	1,100	0.05,0.31	1.14,2.18	5.03,22.94
Lousy Creek	1,700	0.05,0.17	2.87,3.28	2.92,6.83
Lohn Creek	700	0.03,0.11	1.59,1.70	1.79,4.13

* Export coefficients for years 1981 and 1982 are presented before and after a comma, respectively, for each subwatershed and each constituent.

Conclusions

25. The loading of growth-stimulating nutrients is an important determining factor in the trophic state of aquatic ecosystems and the occurrence of problem conditions. Loadings of total phosphorus, total nitrogen, and total organic carbon estimated here are consistent with the current trophic state of Eau Galle Lake and indicative of land use patterns in the surrounding watershed.

26. Documented during this study was the importance of seasonal loading events. Based on data for the Eau Galle River and Lousy Creek, the rapid snowmelt and runoff event of 1982 accounted for 74 to 90 percent of the total phosphorus, 17 to 59 percent of the total nitrogen, and 61 to 79 percent of the total organic carbon loads for that year. The importance of these events, when considered in relation to the rapid flushing of this lake, suggest that seasonal rather than annual loading estimates would provide information of greater value in the interpretation of trophic responses observed during the growing season.

References

- Alberts, E. E., G. E. Schuman, and R. E. Burwell. 1978. Seasonal runoff losses of nitrogen and phosphorus from Missouri Valley loess watersheds. *J. Environ. Qual.* 7:203-207.
- Ashby, S. L., and W. L. James. 1985. Limnology of Eau Galle Tributaries. In R. H. Kennedy, ed. *Limnological studies at Eau Galle Lake, Wisconsin; Report 1, Introduction and water quality monitoring studies.* Technical Report E-85-2. US Army Engineer Waterways Experiment Station, Vicksburg, Miss.
- Beaulac, M. N., and K. H. Beckhow. 1982. An examination of land use-nutrient export relationships. *Wat. Res. Bull.* 18:1013-1024.
- Burwell R. E., D. R. Timmons, and R. F. Holt. 1975. Nutrient transport in surface runoff as influenced by soil cover and seasonal periods. *Soil Sci. Soc. Amer. Proc.* 39:523-528.
- Johnson, D. and G. Lauer. 1985. General Methods. In R. H. Kennedy, ed. *Limnological studies at Eau Galle Lake, Wisconsin; Report 1, Introduction and water quality monitoring studies.* Technical Report E-85-2. US Army Engineer Waterways Experiment Station, Vicksburg, Miss.
- Likens, G. E., and F. H. Bormann. 1974. Linkages between terrestrial and aquatic ecosystems. *Bioscience* 24:447-456.
- Likens, G. E., F. H. Bormann, R. S. Pierce, J. S. Eaton, and N. M. Johnson. 1977. *Biogeochemistry of a forested ecosystem.* Springer-Verlag, New York. 146 pp.
- Reckhow, K. H., and S. C. Chapra. 1983. *Engineering approaches for lake management; Vol 1: Data analysis and empirical modeling.* Butterworth Publishers, Boston. 340 pp.
- Schreiber, J. D., and K. C. McGregor. 1979. The transport and oxygen demand of organic carbon released to runoff from crop residues. *Prog. Wat. Tech.* 2:253-261.
- Timmons, D. R., E. S. Verry, R. E. Burwell, and R. F. Holt. 1977. Nutrient transport in surface runoff and interflow from an aspen-birch forest. *J. Environ. Qual.* 6:188-192.
- Vollenweider, R. A. 1975. Input-output models with special reference to the phosphorus loading concept in limnology. *Schweiz. Z. Hydrol.* 37:53-84.
- Walker, W. W. 1985. *Empirical methods for predicting eutrophication in impoundments; Report 3, Phase II: Model refinements.* Technical Report E-81-9. US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Walker, W. W. 1986. Empirical methods for predicting eutrophication in impoundments; Report 4, Phase III: Applications manual. Technical Report E-81-9. US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

PART III: SEASONAL GROWTH AND COMMUNITY COMPOSITION OF PHYTOPLANKTON*

Introduction

27. Phytoplankton communities have been investigated extensively with attention to environmental factors affecting growth and population dynamics. A great deal of research has been focused on the role of limiting nutrients (Tilman, Kilham, and Kilham 1982) with major emphasis on phosphorus (Schindler 1978). The importance of physical factors has also been recognized (Hickman 1979, Reynolds et al. 1983). It is difficult to distinguish between the effects of chemical and physical factors when dealing with natural phytoplankton communities (Jones 1977). Accordingly, progress in understanding and in predicting the biomass and succession of phytoplankton has been slow, and an experimental approach to phytoplankton ecology has been encouraged (Kalff and Knoechel 1978).

28. In this section the seasonal growth and community composition of phytoplankton in Eau Galle Lake are presented along with an assessment of selected aspects of the chemical and physical environment. The investigation was conducted to establish a base for subsequent experimental work with the phytoplankton community, which is reported in Barko et al. 1986.

Methods and Materials

29. Epilimnetic water samples were collected approximately every 2 weeks during 1981 for phytoplankton identification and enumeration. From preliminary survey analyses, six sampling stations, spanning a depth range of approximately 1 to 9 m, were selected as being collectively representative of the lake. Integral water samples were taken using a tubular collecting device, constructed from a 7.6-cm

* Part III was written by John W. Barko. Derrick J. Bates, Gerald J. Filbin, Susan M. Hennington, and Dwilette G. McFarland contributed technical support.

(3-in.)-diam plastic pipe and equipped with a one-way check valve. The check valve allowed filling of the pipe during vertical immersion and sample retention upon withdrawal from the water. Samples were taken to a depth of 3 m or less, either at shallower stations or occasionally at all stations when the epilimnion became compressed during periods of midsummer calm. Subsamples (125 ml) were preserved using acid-Lugol's solution (Vollenweider 1969), then stored in amber glass bottles at room temperature until processed (usually less than 2 months after collection).

30. Taxonomic identification and enumeration of phytoplankton were accomplished using an inverted microscope (Wild, phase contrast) according to Lund, Kipling, and LeCren (1958). Sample volumes for cell settling varied between 1.0 and 50 ml, depending upon phytoplankton density. Cells with primary dimensions $<64\text{ }\mu\text{m}$ were counted under 560X magnification. Larger forms were counted under 280X magnification or less (see below). With some exceptions, counts were accumulated along one to four strips (i.e., transects) across the 25.5-mm-diam of each counting plate. Particularly large forms were counted under 140X magnification, using the entire plate. At each magnification, a minimum of 300 cells were usually counted. Considerably larger counts were obtained when colonial forms were abundant; counts for these were computed as the product of average cell number per colony (high magnification) and colony number per plate (low magnification).

31. Permanent slides of diatoms were prepared using Hyrax medium following digestion in concentrated HNO_3 . These slides were examined using a Bausch and Lomb phase contrast microscope to determine the relative abundance of different diatom taxa enumerated by inverted microscopy.

32. Cellular volume was computed from average cell dimensions determined for 25 randomly selected cells from each sample (Munawar and Munawar 1976). Volume calculations assumed general conformity to the geometry most closely resembling that of specific algal cells (Vollenweider 1969). Cell volume was converted to fresh weight biomass

assuming that the specific gravity of phytoplankton approximates 1.0 (Nauwerck 1963).

33. Chlorophyll a, corrected for phaeopigments (generally very low in concentration), was determined (American Public Health Association (APHA) 1980) at all stations using the integrated samples collected for phytoplankton analysis. With nearly the same frequency, but on different dates than above, gross primary productivity and net productivity were estimated at stations 20, 30, and 50 from oxygen changes determined (APHA 1980) within 300-ml light and dark glass bottles. These were filled and incubated (3 to 4 hr midday) at two to four discrete depths within the epilimnion. Because of the discrepancy in dates, comparisons involving productivity data necessitated monthly averaging.

34. Various forms of inorganic nitrogen and phosphorus and dissolved silica were determined from water samples collected at two to four discrete depths (all stations) on the same dates as phytoplankton sampling. Associated analyses were conducted colorimetrically using automated (Technicon Auto-Analyzer II) procedures. Vertical profiles of water temperature (all stations) were determined in situ each week using a Hydrolab Surveyor Telethermometer. Water column stability was calculated according to Hutchinson (1957) using the temperature data obtained at station 20, the deepest point in the lake.

35. Variations with depth in the epilimnion among data obtained by discrete sampling were generally minor. These data were, therefore, depth-averaged to allow comparisons with data obtained by integral epilimnetic sampling. Computer-assisted data manipulations and associated statistical analyses were conducted using the Statistical Analysis System (Raleigh, N. C.). All statements of significance made in the text refer to the 5-percent level or less of statistical confidence.

Results

36. In the study, 137 taxa were identified (to species or otherwise) as representatives of seven algal divisions. Approximately 94 percent of total algal biomass and 87 percent of total algal cell

abundance could be accounted for by considering those taxa (38 in total) which individually contributed greater than 2 percent to respective division biomass (Table 5). All divisions were strongly dominated (≥ 75 percent of division biomass) by three or fewer algal taxa. Three species alone (*Stephanodiscus hantzschii*, *Aphanizomenon flos-aquae*, and *Ceratium hirundinella*) collectively contributed about 50 percent of total annual phytoplankton biomass.

37. On the whole, the relationship between algal biomass and abundance was very poor (Table 5) due to broad differences among taxa in cell dimensions and estimated volumes. During bloom periods this relationship improved considerably, however. Notably, two major bloom-forming divisions, Bacillariophyta and Cyanophyta, demonstrated significant linear relationships annually between abundance and biomass with associated R^2 values of 0.91 and 0.61, respectively.

38. Biomass and the seasonal succession of phytoplankton were quite similar at stations 10, 20, 30, and 60 (Figure 7), all of which were located predominantly in pelagic locations. Phytoplankton response at station 40, located at the mouth of the primary inflowing river, and at station 50, located in a shallow (<1-m-deep) littoral area, differed markedly from the former. These stations supported relatively greater Chrysophyte populations beneath the ice in February and March. In addition, both stations, but in particular station 50, demonstrated periodically depressed levels of phytoplankton biomass compared to pelagic stations. At station 40, spring and fall diatom populations were relatively reduced, and at station 50, midsummer Cyanophyte populations were relatively reduced.

39. Considering the lake as a whole, diatoms demonstrated spring and fall dominance with greatest biomass achieved during the spring. The spring diatom population consisted almost exclusively of centric taxa, with *Stephanodiscus hantzschii* and *Melosira* being most prominent. In contrast, centric diatoms were virtually absent during the fall, when pennate taxa (primarily *Asterionella formosa* and *Fragilaria crotonensis*) were prevalent. Cyanophytes, in combination with Dinoflagellates, dominated during the summer months. Cyanophyte biomass peaks, occurring in

Table 5

Relative Biomass and Abundance of Algal Divisions and Associated Taxa*

Division and Taxa**	Biomass, %		Abundance, %	
	Of Division	Of Total	Of Division	Of Total
Bacillariophyta				
<i>Stephanodiscus hantzschii</i>	65.3	30.3	47.8	5.1
<i>Asterionella formosa</i>	11.7	5.4	16.3	1.7
Centrales (unidentified)	7.5	3.5	16.9	1.8
<i>Fragilaria crotonensis</i>	5.0	2.3	4.9	0.5
Pennales (unidentified)	2.4	1.1	3.4	0.4
<i>Melosira</i> sp.	2.1	1.0	1.8	0.2
	94.0	43.6	90.8	9.7
Cyanophyta				
<i>Aphanizomeon flos-aquae</i>	45.3	11.8	33.3	24.4
<i>Anabaena planctonica</i>	24.4	6.3	7.1	5.2
<i>Oscillatoria</i> sp.	10.6	2.8	30.2	22.1
<i>Oscillatoria agardhii</i>	8.0	2.1	16.6	12.1
<i>Anabaena</i> sp.	5.0	1.3	3.7	2.7
Unidentified	2.0	0.5	4.1	3.0
	95.3	24.8	95.0	69.5
Pyrrhophyta				
<i>Ceratium hirundinella</i>	87.0	11.1	30.0	0.1
<i>Glenodinium</i> sp.	9.6	1.2	35.6	0.1
Unidentified	3.4	0.4	34.4	0.1
	100.0	12.7	100.0	0.3
Cryptophyta				
<i>Cryptomonas</i> sp.	43.7	3.0	14.1	0.4
<i>Cryptomonas erosa</i>	28.9	2.0	7.0	0.2
<i>Rhodomonas minuta</i> var. <i>nannoplanctica</i>	7.9	0.5	48.8	1.5

(Continued)

* Taxa included represent those contributing >2 percent to respective division biomass in 1981. Data were derived from samples obtained from six stations throughout the year and are presented as percentage of division (within each division) and percentage of total (across all divisions).

** Divisions and taxa within divisions are arranged in order of decreasing biomass, as percentage of total biomass. Unidentified taxa potentially represent several taxonomic entities.

Table 5 (Concluded)

Division and Taxa **	Biomass, %		Abundance, %	
	Of Division	Of Total	Of Division	Of Total
Cryptophyta (Cont.)				
<i>Rhodomonas minuta</i>	6.8	0.5	19.8	0.6
<i>Cryptomonas marssonii</i>	6.1	0.4	3.3	0.1
Unidentified	4.8	0.3	3.6	0.1
	98.2	6.7	96.6	2.9
Chrysophyta				
Unidentified	47.7	2.0	35.2	1.0
<i>Mallomonas</i> sp.	18.4	0.8	4.0	0.1
<i>Ochromonas</i> sp.	18.0	0.8	13.8	0.4
<i>Chrysochromulina parva</i>	8.8	0.4	38.5	1.1
<i>Chrysococcus</i> sp.	2.5	0.1	0.8	0.0
	95.4	4.1	92.3	2.6
Chlorophyta				
Unidentified	28.2	0.4	22.8	0.7
<i>Chlamydomonas</i> sp.	26.1	0.4	4.7	0.2
<i>Spirogyra</i> sp.	7.5	0.1	0.0	0.0
<i>Ankistrodesmus falcatus</i>	7.3	0.1	1.2	0.0
var. <i>mirabilis</i>				
<i>Staurastrum</i> sp.	5.5	0.1	0.0	0.0
<i>Pediastrum duplex</i>	4.9	0.1	1.3	0.0
var. <i>clathratum</i>				
<i>Dictyosphaerium</i>	3.5	0.1	6.9	0.2
ehrenbergianum				
<i>Scenedesmus</i> sp.	2.4	0.0	5.1	0.2
	85.4	1.3	42.0	1.3
Euglenophyta				
<i>Trachelomonas</i> sp.	52.2	0.4	51.0	0.0
<i>Euglena</i> sp.	35.5	0.2	29.2	0.0
Unidentified	12.3	0.1	19.8	0.0
	100.0	0.7	100.0	0.0

mid-July and mid-September, consisted primarily of heterocyst-bearing *Aphanizomenon flos-aquae* and *Anabaena planctonica*, respectively. Intermittently during the summer, *Ceratium hirundinella* dominated the phytoplankton.

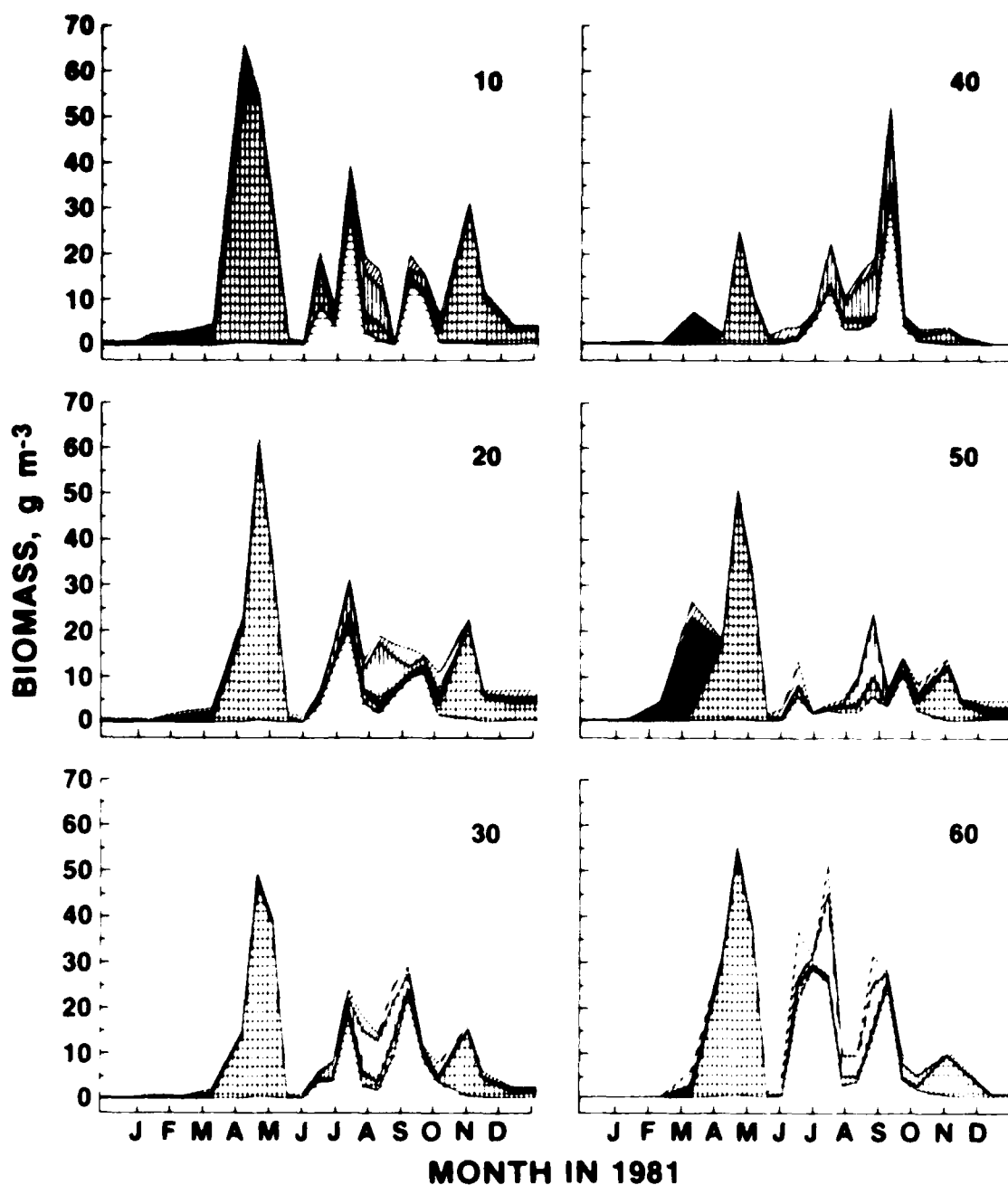


Figure 7. Seasonal succession and biomass (grams per cubic meter) of phytoplankton in the epilimnion at six stations. Stations are identified by number in subfigures. Separately designated divisions include Cyanophyta (□), Bacillariophyta (▨), Chrysophyta (■), and Pyrrhophyta (▤). Separately indistinguishable divisions are grouped in "Other" (▥). Month identifiers designate first day of each month

40. Chlorophyll a provided an excellent indication of phytoplankton biomass except at station 50, where macrophyte and associated epiphyte debris apparently interfered with productivity as well as chlorophyll determinations. Averaged across all stations (excluding station 50), chlorophyll a and biomass were significantly linearly related ($R^2 = 0.8$) (Figure 8). From the slope of the regression line, chlorophyll a in phytoplankton on an average annual lakewide basis can be estimated at 2.8 mg/g fresh mass.

41. Gross primary production and production to biomass (P:B) demonstrated seasonal trends similar to biomass under ice-free conditions (Figure 9). To a limited extent, however, changes in P:B preceded changes in biomass. The P:B varied considerably with both season and associated phytoplankton community composition. Maxima P:B occurred in early summer, while Cyanophytes were dominant, then declined progressively into the fall; minima occurred in the spring and fall during periods of diatom dominance. Net production was significantly linearly correlated ($R^2 = 0.96$) with gross primary production and represented approximately 75 ± 5 percent (standard error) of the latter during the ice-free period.

42. Seasonal fluctuations in lakewide epilimnetic concentrations of soluble reactive phosphorus (SRP), dissolved inorganic nitrogen (DIN), and dissolved silica (SiO_2) were substantial (Figure 10). Concentrations of these nutrients were greatest in the early spring, due to a combination of internal regeneration (primarily DIN and SiO_2) and riverine loading (primarily SRP) processes. Subsequently, in association with spring diatom growth, nutrient concentrations declined rapidly. During the remainder of the year, SRP concentrations approached the lower limit of analytical detection ($5 \mu\text{g/l}$) and appeared to be little influenced by phytoplankton. Following the collapse of the spring diatom population, DIN remained depressed until the lake destratified in the fall. Silica concentrations were depressed by the development of both spring and fall diatom populations, but increased during the summer while diatoms were absent.

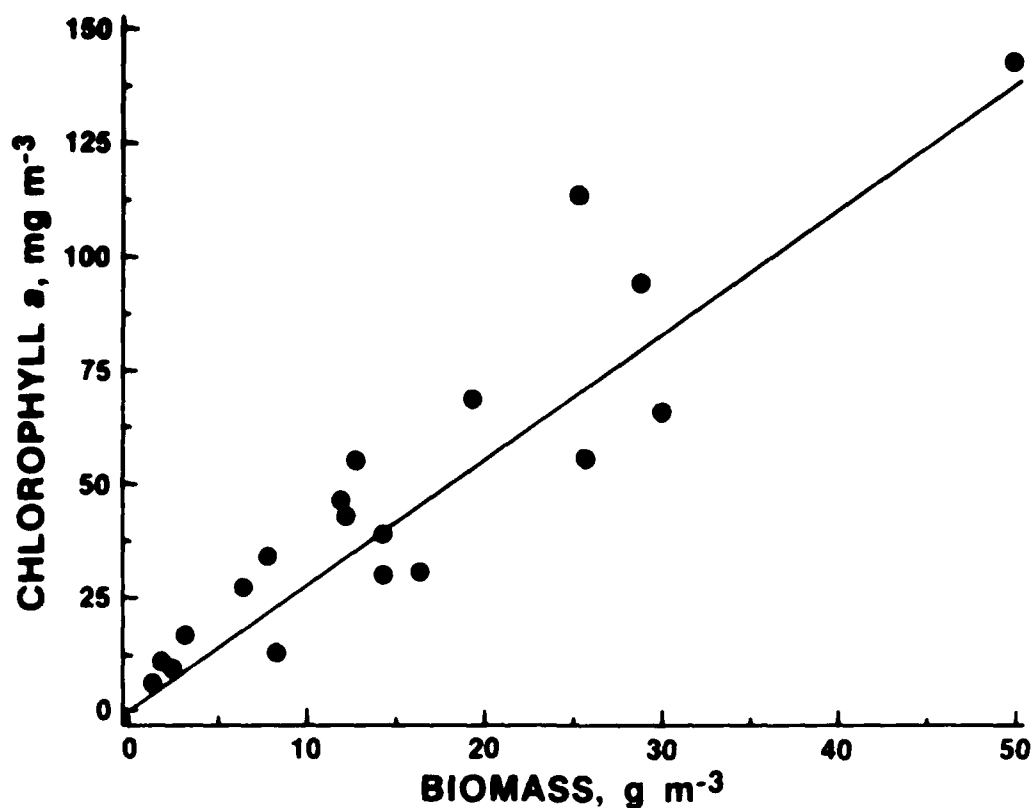


Figure 8. Relationship between epilimnetic chlorophyll a concentration (milligrams per cubic meter) and phytoplankton biomass (grams per cubic meter). Data are lake-wide averages (all stations excluding 50, see text) and are representative of the entire year. Chlorophyll a = $2.77 \times \text{biomass}$; $R^2 = 0.84$; $p < 0.001$

43. Epilimnetic water temperature and water column stability (i.e., resistance to mixing) generally paralleled one another (Figure 11). However, periodic midsummer mixing had a relatively lesser influence on epilimnetic temperature than on stability. These midsummer events tended to temporarily diminish phytoplankton biomass, presumably through epilimnetic dilution; however *Ceratiwm* appeared to respond positively to midsummer mixing (Figure 7).

44. Following spring thaw, Eau Galle Lake remained destratified until late April when the water temperature reached about 11° C. Thereafter, water column stability increased linearly with increasing water temperature until early July, then decreased linearly with decreasing water temperature until early October. Fall destratification

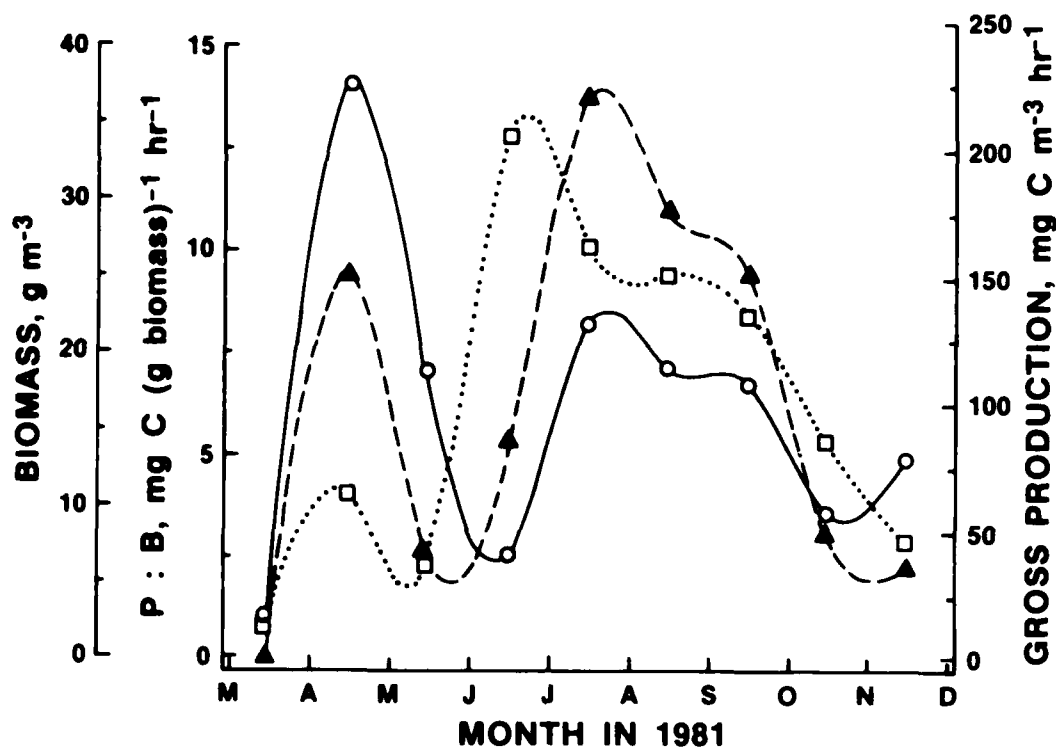


Figure 9. Gross primary production in milligrams carbon per cubic meter per hour ($\text{---}\blacktriangle\text{---}$); phytoplankton biomass in grams per cubic meter ($\text{---}\bigcirc\text{---}$); and production to biomass, P:B, in milligrams carbon per gram biomass per hour ($\text{---}\square\text{---}$) during the ice-free period. Data are monthly epilimnetic averages from data obtained at stations 20 and 30. Month identifiers designate first day of each month

occurred at a water temperature of about 16°C . The collapse of spring diatoms and the resurgence of diatom growth later in the fall (Figure 7) coincided very closely with the onset of summer stratification and fall destratification, respectively (Figure 11).

Discussion

45. The seasonal succession of phytoplankton in Eau Galle Lake was fundamentally similar to that described for temperate eutrophic systems (Hutchinson 1967, Wetzel 1975, Round 1981). Characteristic spring and fall diatom blooms are fairly similar from year to year in these systems. The dominant diatom genera of Eau Galle, *Stephanodiscus* and

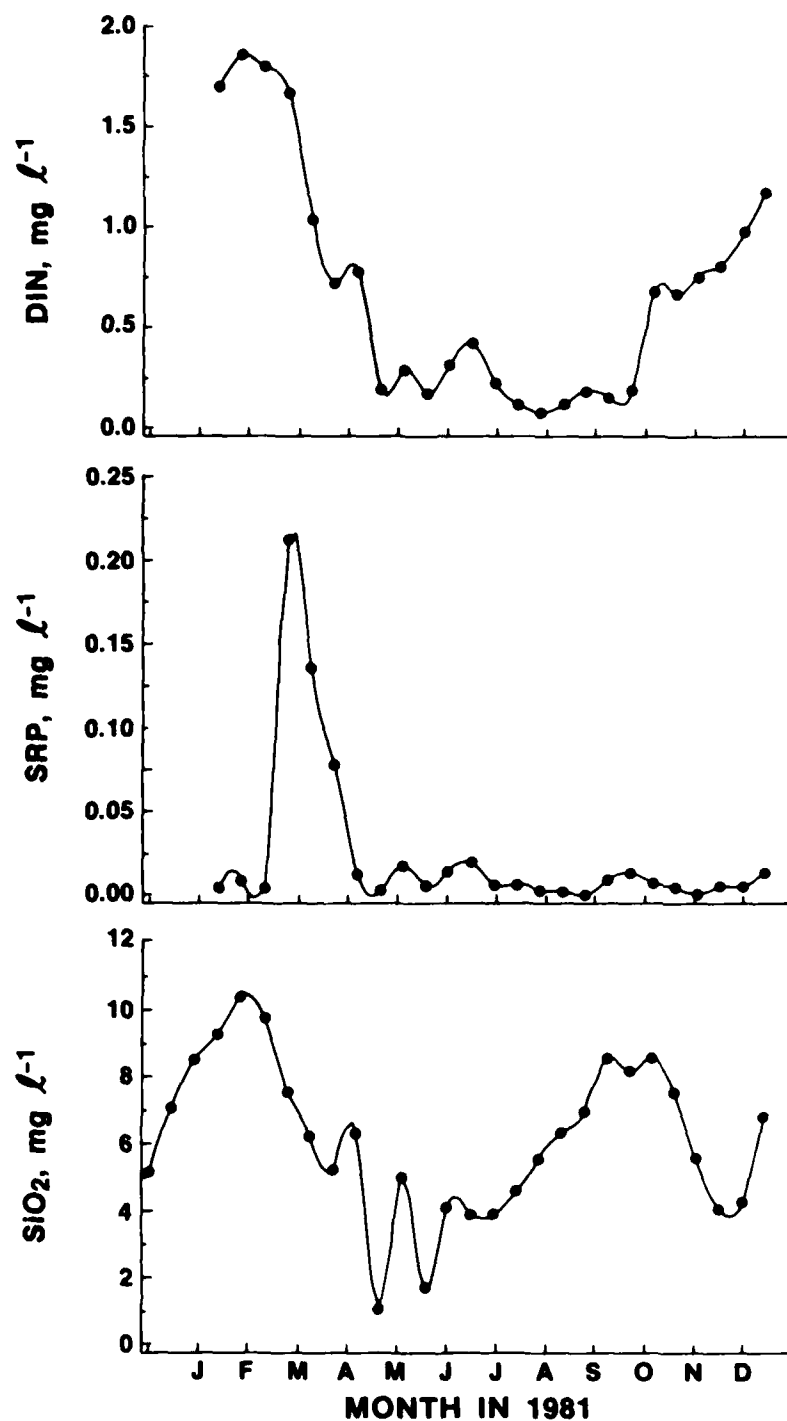


Figure 10. Seasonal variations in DIN, SRP, and SiO₂ (in milligrams per liter). Data are lakewide epilimnetic averages. Month identifiers designate first day of each month

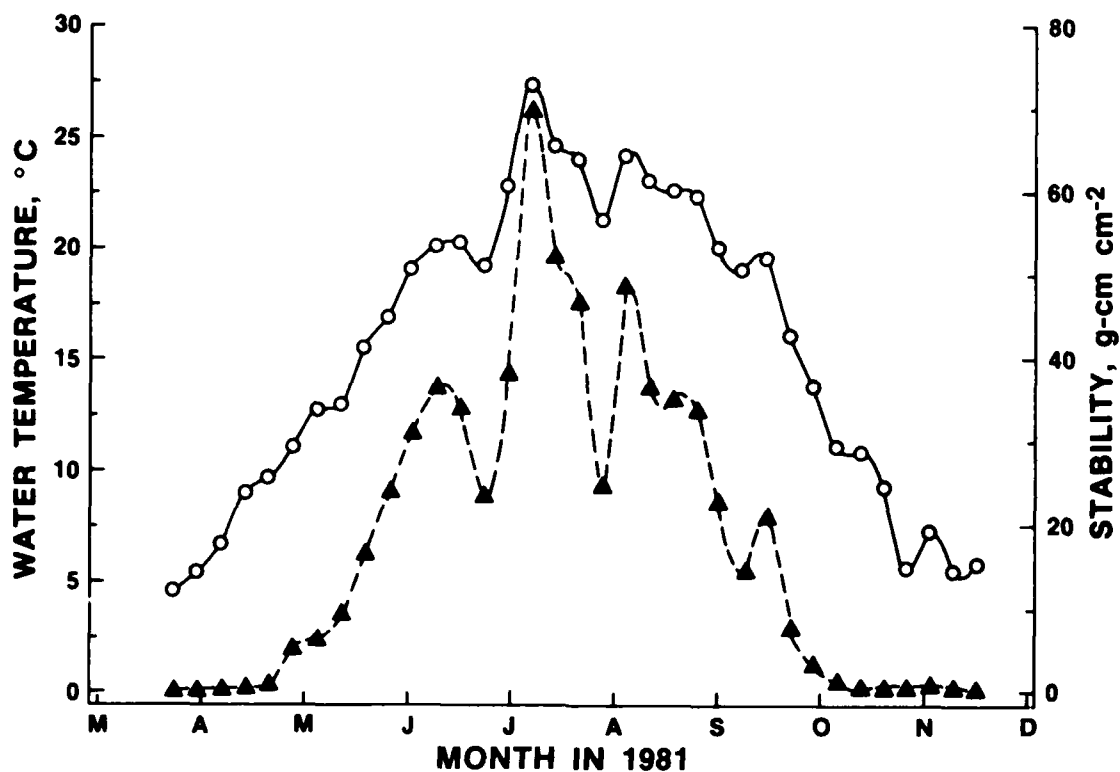


Figure 11. Water column stability in gram-centimeters per square centimeter (---▲---) and lakewide epilimnetic water temperature in degrees Centigrade (—○—). Month identifiers designate first day of each month

Melosira in the spring and *Asterionella* and *Fragilaria* in the fall, collectively represent a common association in fertile lakes (Hutchinson 1967). The significance in Eau Galle of exclusive dominance in the spring by centric diatoms and in the fall by pennate diatoms is not clear; however, this pattern is annually consistent in Eau Galle.

46. Midsummer phytoplankton community composition in productive lakes is quite variable, since Cyanophytes, Chlorophytes, Dinoflagellates, or combinations of the three may prevail (Round 1981). The occurrence in eutrophic systems of substantial dinoflagellate populations (particularly *Ceratium*) is not unusual (Talling 1971, Heaney 1976); however, lakes dominated primarily by cyanophytes have apparently received greater attention (Kalff and Knoechel 1978). As implied in the study of Nicholls, Kennedy, and Hammett (1980), and perhaps applicable

to Eau Galle, competition between *Ceratiwm* and nitrogen-fixing Cyano phytes may involve inorganic nitrogen availability.

47. Phytoplankton diversity is commonly viewed as being lower in eutrophic systems than in oligotrophic systems (Fogg 1980); however, there is little evidence for this. Kalff and Knoechel (1978), in criticizing this view, demonstrated that among lakes ranging broadly in size, location, and trophic state, no more than half a dozen species accounted for the major portion of phytoplankton biomass in each. In Eau Galle, the phytoplankton community was dominated by only three species, each a representative of a different taxonomic division. There is some evidence that dominant phytoplankton species in tropical lakes are no more likely to come from one division than another (Lewis 1978). However, in Eau Galle as in other temperate eutrophic systems, dominants appear to most commonly represent the divisions Bacillariophyta, Cyanophyta, and Pyrrhophyta (Wetzel 1975).

48. On the whole, but with previously noted exceptions, variations in the seasonal succession and biomass of phytoplankton among different sampling stations in Eau Galle were minor. Such uniformity undoubtedly reflects the small size of this particular system and certainly cannot be extended to reservoirs in general (cf. Jasper et al. 1983 and literature cited therein). Reduced phytoplankton biomass during the spring and fall at station 40, near the mouth of the Eau Galle River, suggests dilution or dispersion, since at these times the river and lake were thermally homogeneous. Reduced phytoplankton biomass during the summer at station 50 in the littoral zone related negatively to the development of the submersed macrophyte community (Filbin and Barko 1985). Negative associations between submersed macrophytes and phytoplankton have been reported elsewhere (Brammer 1979 and literature cited therein) and appear to reflect primarily competition for nutrients, although the involvement of allelochemical substances (Whittaker and Feeny 1971, Wium-Anderson, Christophersen, and Houen 1982) is also possible.

49. Phytoplankton biomass and chlorophyll a were closely related throughout the year in Eau Galle. Estimated phytoplankton chlorophyll a

content (2.8 mg/g fresh mass) compares favorably with physiological estimates (e.g., Verity 1981). Thus, the two methods of algal biomass estimation (inverted microscopy and chlorophyll analysis) appear to have provided similar results. Excellent correspondence between phytoplankton biomass and chlorophyll a has been reported elsewhere (e.g., Bailey-Watts 1982) but interpreted with caution. In the investigation of Jones (1977) a pronounced discrepancy between biomass and chlorophyll a was attributed to higher chlorophyll per unit cell volume in diatoms than in blue-green algae. Within single species, nutrients, light, and water temperature have been shown to interact in influencing phytoplankton chlorophyll a content (Rhee and Gotham 1981a,b). Thus, it is perhaps reasonable to expect chlorophyll a and biomass responses to be uncoupled more often than not. The converse situation in Eau Galle might reflect the relative invariance in light penetration during the growing season (mean secchi depth = 1.12 ± 0.16 m (standard error)), which possibly reduced seasonal variance in cellular chlorophyll a content.

50. The relatively low ratio of production to biomass (P:B) during the spring diatom bloom suggests a low cellular loss rate, a high efficiency of biomass yield, or both at that time. Higher P:B during the summer than during either the spring or fall suggests least efficient maintenance of phytoplankton biomass in the epilimnion during the summer. Since respiration accounted for nearly a constant fraction of gross production during the growing season, seasonal variations in nonrespiratory losses (e.g., sinking, grazing, parasitism, etc.; see Jassby and Goldman 1974, Knoechel and Kalff 1978, Crumpton and Wetzel 1982) likely accounted for related variations in P:B. Changes in species composition have been suggested to influence the relationship between production and biomass in phytoplankton communities (Berman and Pollinger 1974, Munawar and Munawar 1982), but physical and chemical factors are probably also involved.

51. Phytoplankton biomass in Eau Galle on the whole was generally comparable to that reported for other eutrophic systems in temperate regions of the world (cf. Kalff and Knoechel 1978). However, peak diatom biomass (lakewide average ca. 50 g/m^3 in the spring) was

exceptionally high, nearly twice that predicted from the silica concentration preceding the spring bloom (refer to Figure 10 in Kalff and Knoechel 1978). Thus, factors in addition to silica concentration (perhaps rapid silica flux) apparently enhanced the development of the spring diatom population.

52. Numerous studies, including the present, have demonstrated reductions in dissolved silica concentration with the growth of diatoms. Whereas silica availability appears to dictate maximum obtainable diatom biomass under some circumstances (Lund 1950; Bailey-Watts 1976; Jewson, Rippey, and Gilmore 1981), the minimum concentration of silica required to sustain diatoms is not precisely known (Round 1981). Silica requirements probably vary among diatom species (Mechling and Kilham 1982, Sommer and Stabel 1983). With the exception of two sampling dates during late spring, lakewide average dissolved silica concentrations in Eau Galle never fell below 4 mg/l (SiO_2), which is probably above the minimum usable concentration for diatoms. Depletion of silica during the spring and fall may have reduced diatom growth, but is unlikely to have contributed prominently to the collapse of respective diatom populations.

53. Among the chemical variables considered here, DIN potentially had the greatest influence on the seasonal succession of phytoplankton in Eau Galle. The occurrence of diatom populations, both spring and fall, was directly associated with maximum DIN concentration; nitrogen-fixing Cyanophyte populations persisted during the summer while DIN remained low. Experimental additions of DIN have been demonstrated to reduce the magnitude of *Aphanizomenon* blooms and promote changes in phytoplankton species composition (Barica, Kling, and Gibson 1980).

54. The relative proportion of Cyanophytes in the epilimnetic phytoplankton of lakes may depend on the ratio of total nitrogen to total phosphorus (TN:TP) in the lake water; Cyanophytes tend to dominate in lakes where TN:TP is 29 on a mass basis (Smith 1983). In Eau Galle, TN:TP during the ice-free period (1981) was 16 ± 1.2 (standard error), suggesting a propensity for Cyanophyte occurrence. Nitrogen deficits may stimulate the development of nitrogen-fixing Cyanophyte species,

which are capable of rectifying nutrient imbalances (Schindler 1977). Massive inputs of phosphorus relative to nitrogen (TN:TP ca. 10) during the 1981 spring thaw clearly established a potential nutrient imbalance in Eau Galle.

55. Often in eutrophic systems the most troublesome phytoplankton species become dominant at about the same time that nutrient concentrations are minimal. Ambient nutrient concentrations clearly do not measure availability, since related fluxes and the content of nutrients in algal cells are ignored (Reynolds and Walsby 1975). Eau Galle is periodically mixed during the summer, as evidenced by reductions in water column stability. These events undoubtedly transport nutrients from bottom waters upward into the epilimnion. *Ceratium* may benefit from associated increases in DIN availability (Nicholls, Kennedy, and Hammett 1980). Since Cyanophytes among other algae are effective at luxuriously concentrating excess nitrogen and phosphorus (Gerloff and Skoog 1954), it is not surprising that soluble concentrations of these elements in the epilimnion were little influenced by midsummer mixing.

56. In shallow systems, such as Eau Galle Lake, physical factors may ultimately be more important than chemical factors in affecting the phytoplankton (Haertel 1976, Hickman 1979). Since the light requirements of Cyanophytes appear fundamentally similar to those of other phytoplankton (Reynolds and Walsby 1975), light may be less important than other physical factors contributing to phytoplankton succession. Differences in the temperature requirements of phytoplankton often seem to influence the seasonal succession of particular species (Hutchinson 1967, Patrick 1971). However, it is unlikely that temperature alone explains the abrupt seasonal replacement of diatom populations by Cyanophytes or other algae in temperate lakes, since diatoms as well as other algae demonstrate broad ranges in thermal tolerance (Round 1968, Patrick 1977). Evidence is accumulating which suggests that shifts in the species composition of phytoplankton communities may occur in response to major changes in water column stability rather than temperature *per se* (Reynolds et al. 1983 and literature cited therein).

57. Diatoms are generally subject to high loss rates due to sinking, because they are relatively dense and lack the ability to regulate their position within the water column (Hutchinson 1967). For this reason diatoms are obviously disadvantaged in hydraulically stable environments; this may explain their absence from the phytoplankton community in Eau Galle during the summer. Diatom populations can be maintained through artificial mixing (Lund 1971, Reynolds et al. 1983), although adequate nutrition (particularly silica) is presupposed. The Cyanophyte and *Ceratium* populations in Eau Galle, as elsewhere, are undoubtedly benefited during stratified periods by their respective capabilities of buoyancy regulation (Reynolds and Walsby 1975) and mobility (George and Heaney 1978).

58. In a restricted sense, phytoplankton succession in temperate eutrophic systems involves diatoms → summer phytoplankton → diatoms. In Eau Galle this successional sequence appears to be closely related to changes in water column stability and inorganic nitrogen availability. Considering the abundance of silica in this system, it is possible that prolonged vernal mixing could extend diatom dominance into the summer. Alternatively, an increase in the ratio of TN to TP or an increase in DIN availability might decrease the proportion of nitrogen-fixing Cyanophytes with respect to total phytoplankton.

References

- American Public Health Association. 1980. Standard methods for the examination of water and wastewater, 15th ed. New York.
- Bailey-Watts, A. E. 1976. Planktonic diatoms and some diatom-silica relations in a shallow eutrophic Scottish loch. *Freshwater Biol.* 6:69-80.
- _____. 1982. The composition and abundance of phytoplankton in Loch Leven (Scotland), 1977-1979, and a comparison with the succession in earlier years. *Int. Revue ges Hydrobiol.* 67:1-25.
- Barica, J., H. Kling, and J. Gibson. 1980. Experimental manipulation of algal bloom composition by nitrogen addition. *Can. J. Fish. Aquat. Sci.* 37:1175-1183.

- Barko, J. W., A. R. Klemmer, D. G. McFarland, and M. S. Hennington. 1986. Experimental manipulations of phytoplankton in a eutrophic impoundment. Miscellaneous Paper in preparation. US Army Engineer Waterways Experiment Station, Vicksburg, Miss.
- Berman, T., and U. Pollinger. 1974. Annual and seasonal variations of phytoplankton, chlorophyll, and photosynthesis in Lake Kinneret. *Limnol. Oceanogr.* 19:31-54.
- Brammer, E. S. 1979. Exclusion of phytoplankton in the proximity of dominant water-soldier (*Stratiotes aloides*). *Freshwater Biol.* 9:233-249.
- Crumpton, W. G., and R. G. Wetzel. 1982. Effects of differential growth and mortality in the seasonal succession of phytoplankton populations in Lawrence Lake, Michigan. *Ecology* 63:1729-1739.
- Filbin, G. J., and J. W. Barko. 1985. Growth and nutrition of submersed macrophytes in a eutrophic Wisconsin impoundment. *Freshwater Ecology* 3:275-285.
- Fogg, G. E. 1980. Phytoplankton primary production. Pages 24-45 in R. S. K. Barnes and K. H. Mann, eds. *Fundamentals of aquatic ecosystems*. Blackwell Scientific Publications, Oxford.
- George, D. G., and S. I. Heaney. 1978. Factors influencing the spatial distribution of phytoplankton in a small productive lake. *J. Ecol.* 66:133-155.
- Gerloff, G. C., and F. Skoog. 1954. Cell contents of nitrogen and phosphorus as measure of the availability for growth of *Microcystis aeruginosa*. *Ecology* 35:348-353.
- Haertel, L. 1976. Nutrient limitation of algal standing crops in shallow prairie lakes. *Ecology* 57:664-678.
- Heaney, S. I. 1976. Temporal and spatial distribution of the dinoflagellate *Ceratium hirundinella* O. F. Muller within a small productive lake. *Freshwater Biol.* 6:531-542.
- Hickman, M. 1979. Phytoplankton production in a small eutrophic lake in central Alberta, Canada. *Int. Revue ges Hydrobiol.* 64:643-659.
- Hutchinson, G. E. 1957. A treatise on limnology; I. Geography, physics, and chemistry. Wiley, New York.
- _____. 1967. A treatise on limnology; II. Introduction to lake biology and the limnoplankton. Wiley, New York.
- Jasper, S., E. C. Carmack, R. J. Daley, C. B. J. Gray, C. H. Pharo, and R. C. Wiegand. 1983. Primary productivity in a large, temperate lake with river inflow: Kootenay Lake, British Columbia. *Can. J. Fish. Aquat. Sci.* 40:319-327.
- Jassby, A. D., and C. R. Goldman. 1974. Loss rates from a lake phytoplankton community. *Limnol. Oceanogr.* 19:618-627.

- Jewson, D. H., B. H. Rippey, and W. K. Gilmore. 1981. Loss rates from sedimentation, parasitism, and grazing during the growth, nutrient limitation, and dormancy of a diatom crop. *Limnol. Oceanogr.* 26:1045-1056.
- Jones, R. I. 1977. Factors controlling phytoplankton production and succession in a highly eutrophic lake (Kennego Bay, Lough Neagh); 1. The phytoplankton community and its environment. *J. Ecol.* 65:547-559.
- Kalff, J., and R. Knoechel. 1978. Phytoplankton and their dynamics in oligotrophic and eutrophic lakes. *Ann. Rev. Ecol. Syst.* 9:475-495.
- Knoechel, R., and J. Kalff. 1978. An in situ study of the productivity and population dynamics of five freshwater planktonic diatom species. *Limnol. Oceanogr.* 23:195-218.
- Lewis, W. M., Jr. 1978. A compositional, phytogeographical and elementary structural analysis of the phytoplankton in a tropical lake: Lake Lanao, Philippines. *J. Ecol.* 66:213-226.
- Lund, J. W. G. 1950. Studies on *Asterionella formosa* Hass; II. Nutrient depletion and the spring maximum. *J. Ecol.* 35:15-35.
- _____. 1971. An artificial alteration of the seasonal cycle of the plankton diatom *Melosira italica* subsp. *subarctica* in an English lake. *J. Ecol.* 59:521-533.
- Lund, J. W. G., C. Kipling, and E. D. LeCren. 1958. The inverted microscope method of estimating algal numbers and the statistical basis of estimations by counting. *Hydrobiol.* 11:143-170.
- Mechling, J. A., and S. S. Kilham. 1982. Temperature effects on silicon limited growth of the Lake Michigan diatom *Stephanodiscus minutus* (Bacillariophyceae). *J. Phycol.* 18:199-205.
- Munawar, M., and I. F. Munawar. 1976. A lakewide study of phytoplankton biomass and its species composition in Lake Erie, April-December 1970. *J. Fish. Res. Board Can.* 33:581-600.
- _____. 1982. Phycological studies in lakes Ontario, Erie, Huron, and Superior. *Can. J. Bot.* 60:1837-1858.
- Nauwerck, A. 1963. The relation between zooplankton and phytoplankton in Lake Erken. *Symb. Bot. Ups.* 17:163. (In German.)
- Nicholls, K. H., W. Kennedy, and C. Hammett. 1980. A fish-kill in Heart Lake, Ontario, associated with the collapse of a massive population of *Ceratium hirundinella* (Dinophyceae). *Freshwater Biol.* 10:553-561.
- Patrick, R. 1971. The effects of increasing light and temperature on the structure of diatom communities. *Limnol. Oceanogr.* 16:405-421.
- _____. 1977. Ecology of freshwater diatoms and diatom communities. Pages 284-332 in D. Werner, ed. *The biology of diatoms*. University of California Press, Berkeley and Los Angeles.
- Reynolds, C. S., and A. E. Walsby. 1975. Waterblooms. *Biol. Rev.* 50:437-481.

- Reynolds, C. S., S. W. Wiseman, B. M. Godfrey, and C. Butterwick. 1983. Some effects of artificial mixing on the dynamics of phytoplankton populations in large limnetic enclosures. *J. Plankton Res.* 5:203-234.
- Rhee, G-Y., and I. J. Gotham. 1981a. The effect of environmental factors on phytoplankton growth: temperature and the interactions of temperature with nutrient limitation. *Limnol. Oceanogr.* 26:635-648.
- _____. 1981b. The effect of environmental factors on phytoplankton growth: light and the interactions of light with nitrate limitation. *Limnol. Oceanogr.* 26:649-659.
- Round, F. E. 1968. Light and temperature. In D. F. Jackson, ed. *Algae, man and the environment*. Syracuse University Press, New York.
- _____. 1981. *The ecology of algae*. Cambridge University Press, Cambridge, Mass.
- Schindler, D. W. 1977. Evolution of phosphorus limitation in lakes: natural mechanisms compensate for deficiencies of nitrogen and carbon in eutrophied lakes. *Science* 195:260-262.
- _____. 1978. Factors regulating phytoplankton production and standing crop in the world's freshwaters. *Limnol. Oceanogr.* 23:478-486.
- Smith, V. H. 1983. Low nitrogen:phosphorus ratios favor dominance by bluegreen algae in lake phytoplankton. *Science* 22:669-671.
- Sommer, U., and H. H. Stabel. 1983. Silicon consumption and population density changes of dominant planktonic diatoms in Lake Constance. *J. Ecol.* 71:119-130.
- Talling, J. F. 1971. The underwater light climate as a controlling factor in the production ecology of freshwater phytoplankton. *Mitt. Int. Verein. Theor. Angew. Limnol.* 19:214-243.
- Tilman, D., S. S. Kilham, and P. Kilham. 1982. Phytoplankton community ecology: the role of limiting nutrients. *Ann. Rev. Ecol. Syst.* 13:349-372.
- Verity, P. G. 1981. Effects of temperature, irradiance, and daylength on the marine diatom *Leptocylindrus danicus* Cleve; I. Photosynthesis and cellular composition. *J. Exp. Mar. Biol. Ecol.* 55:79-91.
- Vollenweider, R. A., ed. 1969. *A manual on methods for measuring primary production in aquatic environments*, 2d ed. IBP Handbook No. 12. Blackwell Scientific Publications, Oxford.
- Wetzel, R. G. 1975. *Limnology*. W. B. Saunders, Philadelphia, Pa.
- Whittaker, R. H., and P. P. Feeny. 1971. Allelochemicals: chemical interactions between species. *Science* 171:757-770.
- Wium-Andersen, S. U. Anthoni, C. Christophersen, and G. Houen. 1982. Allelopathic effects on phytoplankton by substances isolated from aquatic macrophytes (Charales). *Oikos* 39:187-190.

PART IV: THE ZOOPLANKTON COMMUNITY OF EAU GALLE LAKE*

Introduction

59. The zooplankton community is an important component in a lake ecosystem. It provides the link between the primary producers and the upper levels of the food chain. Herbaceous zooplankters are eaten by other zooplankton and other invertebrates as well as by fish. Many fish utilize zooplankton in the early part of their life cycle.

60. The zooplankton community structure may also indicate the trophic status of a water body. Many zooplankton species have specific physiological requirements and thus are only found in certain types of environments.

61. Described here is the zooplankton community of Eau Galle Lake from April 1980 through October 1982. The seasonal trends are discussed as well as factors which regulate the community.

Methods

62. Samples for zooplankton analysis were collected from Eau Galle Lake at stations 20 and 50. Samples were collected by vertical tow with a Wisconsin style plankton net having a 15-cm diameter and using 76- μ mesh netting. Station 20 was located in the deepest portion of the reservoir where the water depth was approximately 9 m. The net was only pulled through the zone of oxygenation. This station was used to characterize the pelagial zone. Station 50 characterized the littoral area of the reservoir. The water depth at this station was generally 0.8 to 1.2 m. Two tows were pulled at each station through the summer of 1981. Starting in September, only one tow was pulled at station 20, but two tows were continued at station 50.

63. Samples were collected by personnel from the Eau Galle Laboratory of the US Army Corps of Engineers. Samples were mailed to the

* Part IV was written by Paul J. Garrison.

author where they were concentrated to the appropriate counting volumes. Generally, three replicates each of 1-ml volume were counted from the samples.

64. An attempt was made to segregate copepod nauplii into calanoids and cyclopoids when counting. However, since data analysis indicates that this was not accurate, the naupliar groups have been combined for this report.

Results

Station 20

65. Of the 48 species of zooplankton netted at station 20, 60 percent were rotifers. The dominant rotifer was *Keratella cochlearis*. *Fomphlox sulcata* and *Polyarthra vulgaris* were occasionally important components of the rotifer community. The dominant copepods were the cyclopoids *Cyclops bicuspidatus thomasi* and *Mesocyclops edax*. The important cladocerans were *Laphnia galeata mendotae*, *Bosmina longirostris*, and *L. retrocurva*, and occasionally *L. ambigua* and *L. parvula*.

66. Figure 12 indicates the seasonal trends of the major zooplankton groups at station 20. The rotifers exhibited the highest numbers, having large peaks which were of short duration in May 1981, late July 1982, and October 1982. During the remainder of the study period, rotifers were present in much lower numbers although they were often more abundant than either of the other zooplankton groups. The copepods were the next most numerically important group, reaching peaks in excess of 160 organisms/liter. Copepods exhibited peaks in May and August 1981 and again in May 1982. The cladocerans exhibited three peaks during the study. Cladocerans peaked both years in June and again in October 1982.

67. The seasonal trend of *Keratella cochlearis* is very similar to the trend of the total rotifer community (Figure 13). *K. cochlearis* exhibited a peak in May 1981, but one was not observed the following May. This may have been caused by competition from *Bosmina* which was not present in the spring of 1981. *Bosmina* utilizes similar food resources as *K. cochlearis*. A peak occurred in October 1982 but a

Station 20

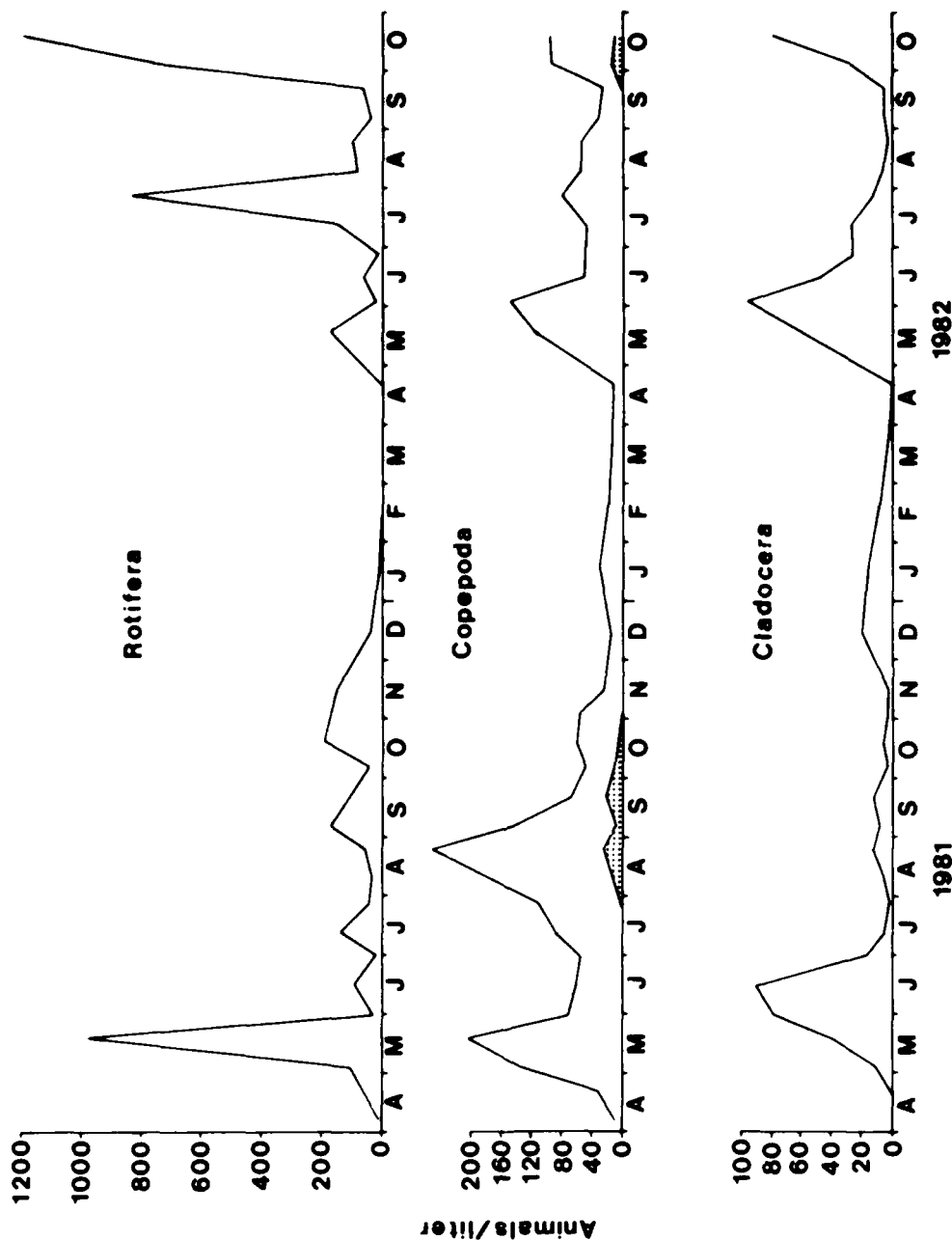


Figure 12. Seasonal trends of major zooplankton groups at station 20. The lower graph of Copepods represents calanoid adults and copepodites

Station 20

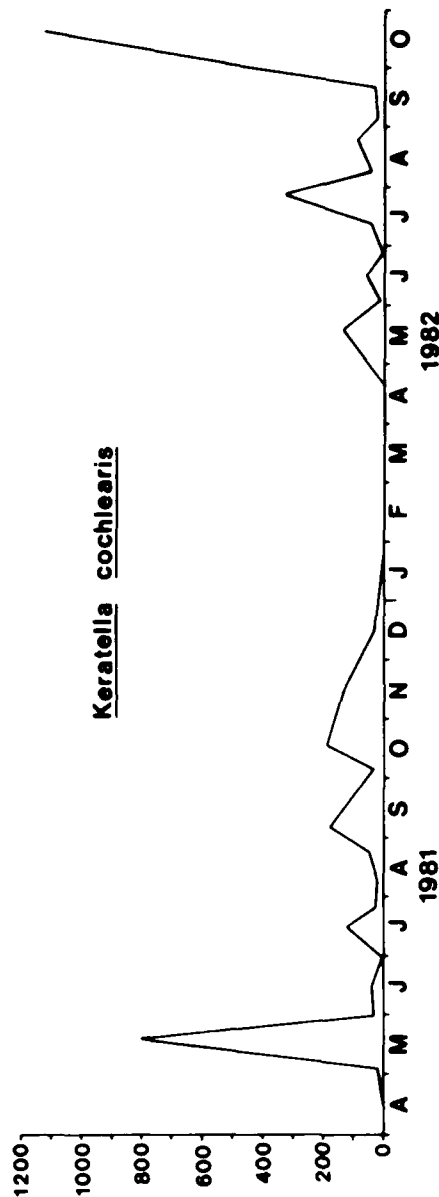
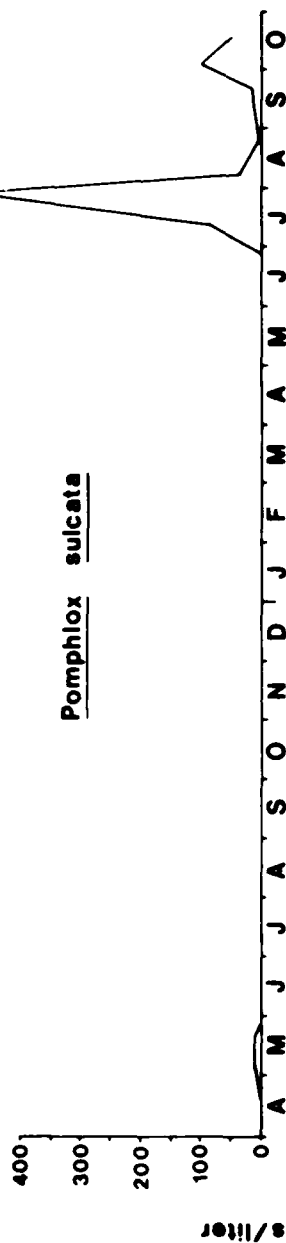
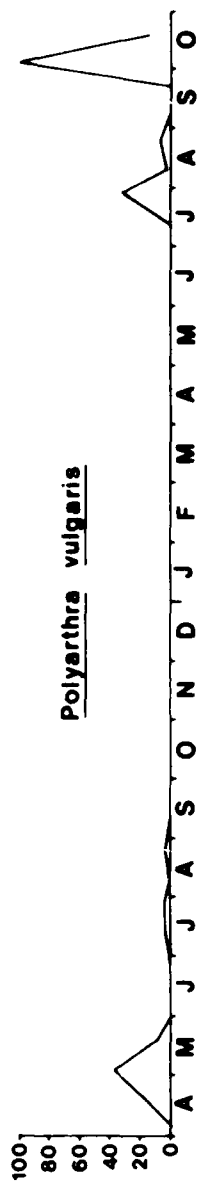


Figure 13. Seasonal changes of dominant rotifers in the pelagial zone. Note the differences in the vertical scales

similar peak did not occur in 1981. Much lower levels were observed the rest of the study period, especially during the ice-covered period.

68. *Pomphlox sulcata* was present only in low numbers in 1981, but a large peak occurred in July 1982 (Figure 13). A smaller peak occurred in early October. The only other rotifer of any consequence, *Polysarthra vulgaris*, was present during the spring of 1981, but did not appear until late July 1982 and reached its greatest numbers in early October 1982.

69. *Mesocyclops edax* comprised the bulk of the copepods during the summers of both years (Figure 14). *M. edax* adults reached similar levels both years although more copepodites occurred in 1982. *Cyclops bicuspidatus thomasi* mainly occurred in the spring and fall, with a smaller population during the ice-covered period. Adults seemed less abundant in 1982 although the opposite was true for the copepodites. Copepodites of *C. b. thomasi* are known to encyst when conditions are unfavorable. This may explain their decline following the spring pulse. The calanoid copepods were represented by *Skistodiaptomus oregonensis* and *Leptodiaptomus siciloides*. These were only present during late summer and fall during both years (Figure 12). Although it was difficult to distinguish between cyclopoid and calanoid nauplii, most were probably cyclopoid. In addition, some of the nauplii observed prior to the increase of diaptomids were probably calanoid nauplii.

70. *Daphnia galeata mendotae* exhibited a peak in early summer for 1981 and 1982 (Figure 15). *D. parvula* exhibited a peak just prior to the 1981 peak of *D. g. mendotae* and again in 1982. *D. retrocurva* exhibited only a minor peak in 1981. *D. ambigua* was present during early summer in 1981, but not in 1982. *Bosmina longirostris* was present only in small numbers prior to December 1981 (Figure 15). A peak occurred in December and lasted into January. In May 1982 *B. longirostris* exhibited a short pulse and increased again in October. *Diaphanosoma leuchtenbergianum* was present in low numbers for both years in late summer.

Station 50

71. Rotifers comprised the majority of species (68%) at station 50. As observed at station 20 the dominant rotifer at station 50 was

Station 20

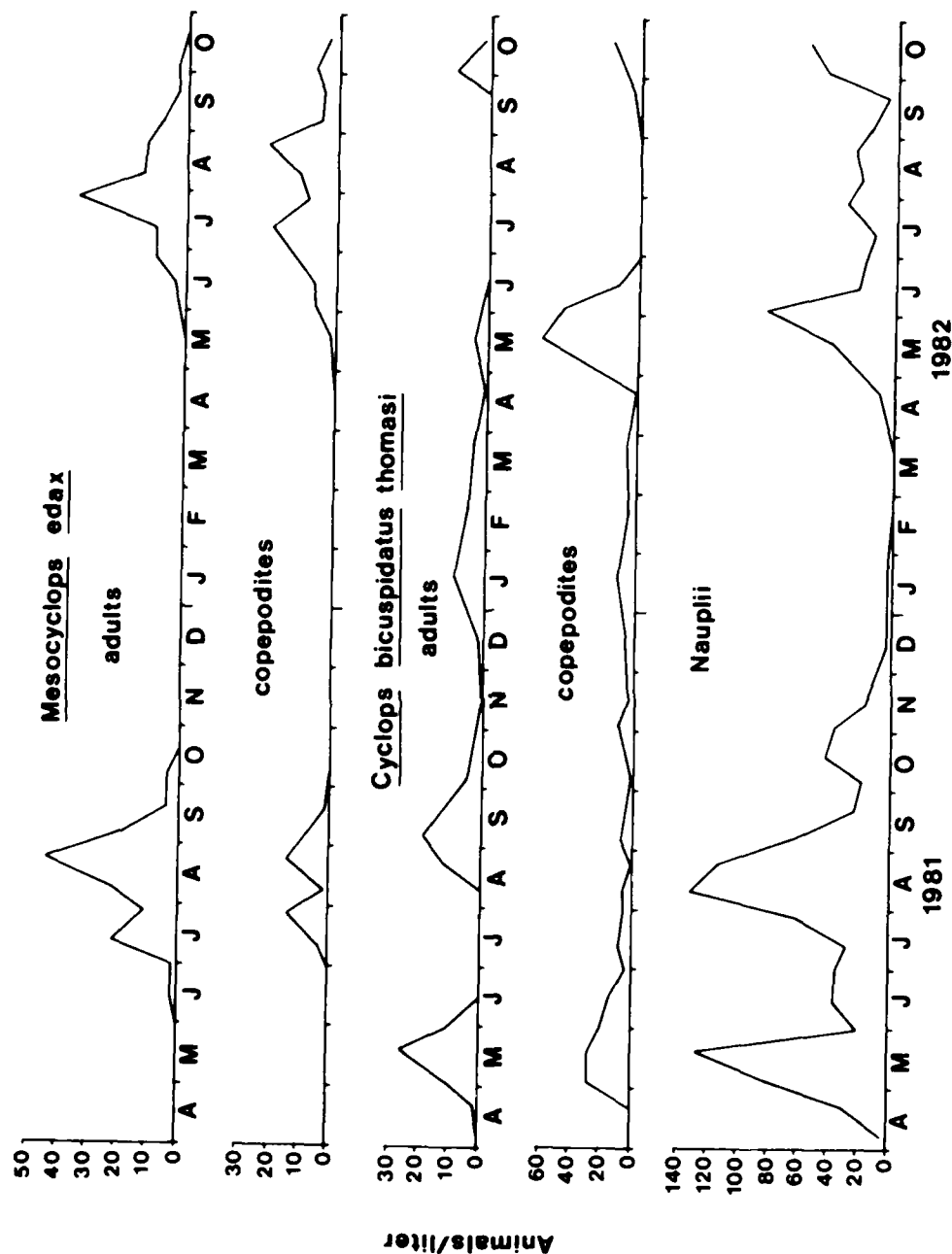


Figure 14. Seasonal trends of the dominant copepods at station 20

Station 20

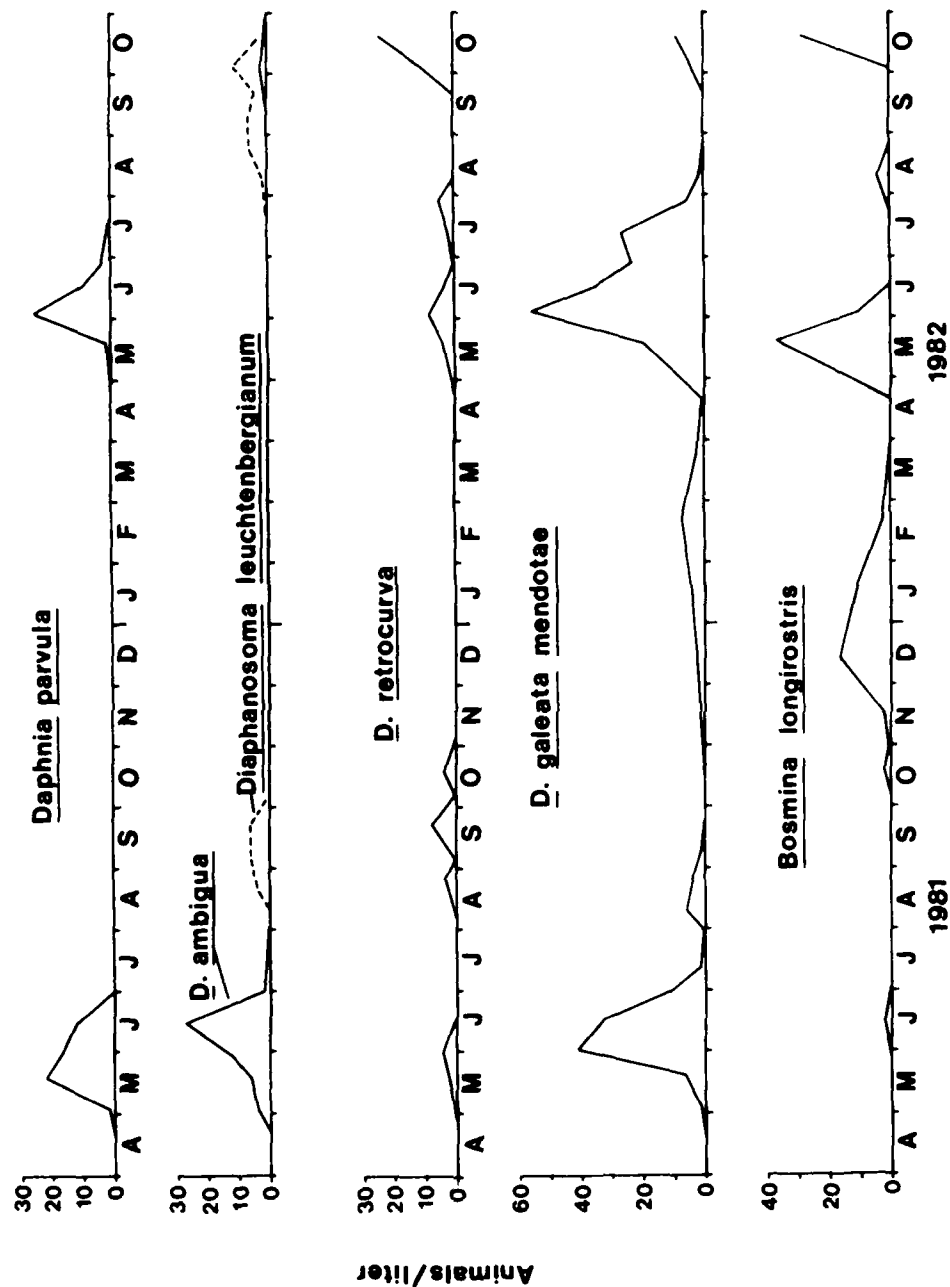


Figure 15. Seasonal changes of the important cladocerans at station 20

Keratella cochlearis. Other rotifers such as *Polyarthra vulgaris*, *P. dolichoptera*, *Synchaeta* sp., and *Pomphlox sulcata* were present in lesser amounts. The dominant copepods were *Cyclops bicuspidatus thomasi* and *Acanthocyclops vernalis*. The dominant cladoceran was *Bosmina longirostris*, although *Daphnia galeata mendota*, *D. ambigua*, *Ceriodaphnia lacustris*, and *Chydorus sphaericus* were occasionally common. Rotifer peaks were observed in May and late November of 1981 and May and October of 1982 (Figure 16). During the remainder of the year, rotifers were present in lower numbers; however, they were often more abundant than the other two zooplankton groups. The copepods were most abundant from May to August both years (Figure 16). Summer peaks were around 150 organisms/liter both years. The cladocerans reached higher numbers than the copepods, especially in 1982. Like the copepods, they were most abundant from May to August.

72. The seasonal trend of the rotifer population is largely described by *Keratella cochlearis* (Figure 17). This rotifer exhibited peaks during spring both years and in late November 1981, and increased in October 1982. The greatest numbers of *K. cochlearis* occurred in late November 1982 when 1,200 organisms/liter were present. *Pomphlox sulcata* was present in spring 1981 and again in July and August of 1982. A smaller peak occurred in early October 1982. In 1981 *Synchaeta* sp. was present during the fall but had a short peak in June 1982. *Polyarthra vulgaris* was present in the spring of 1981, but in 1982 it did not occur until fall (Figure 18). *P. dolichoptera* was present in late summer 1981 and in spring and early summer of 1982.

73. *Acanthocyclops vernalis* was present only during the summer for both years (Figure 19). Although more copepodites were found in 1982, the numbers of adults were similar in both years. Copepodites of *Mesocyclops edax* were more common in 1982, but very few adults were encountered. *Cyclops bicuspidatus thomasi* was usually most abundant in spring and late fall and winter. Very few adults were present in 1982. Although *A. vernalis* and *M. edax* were more common in 1982, *C. b. thomasi* appeared to decline. Very few calanoids were present, so most of the

Station 50

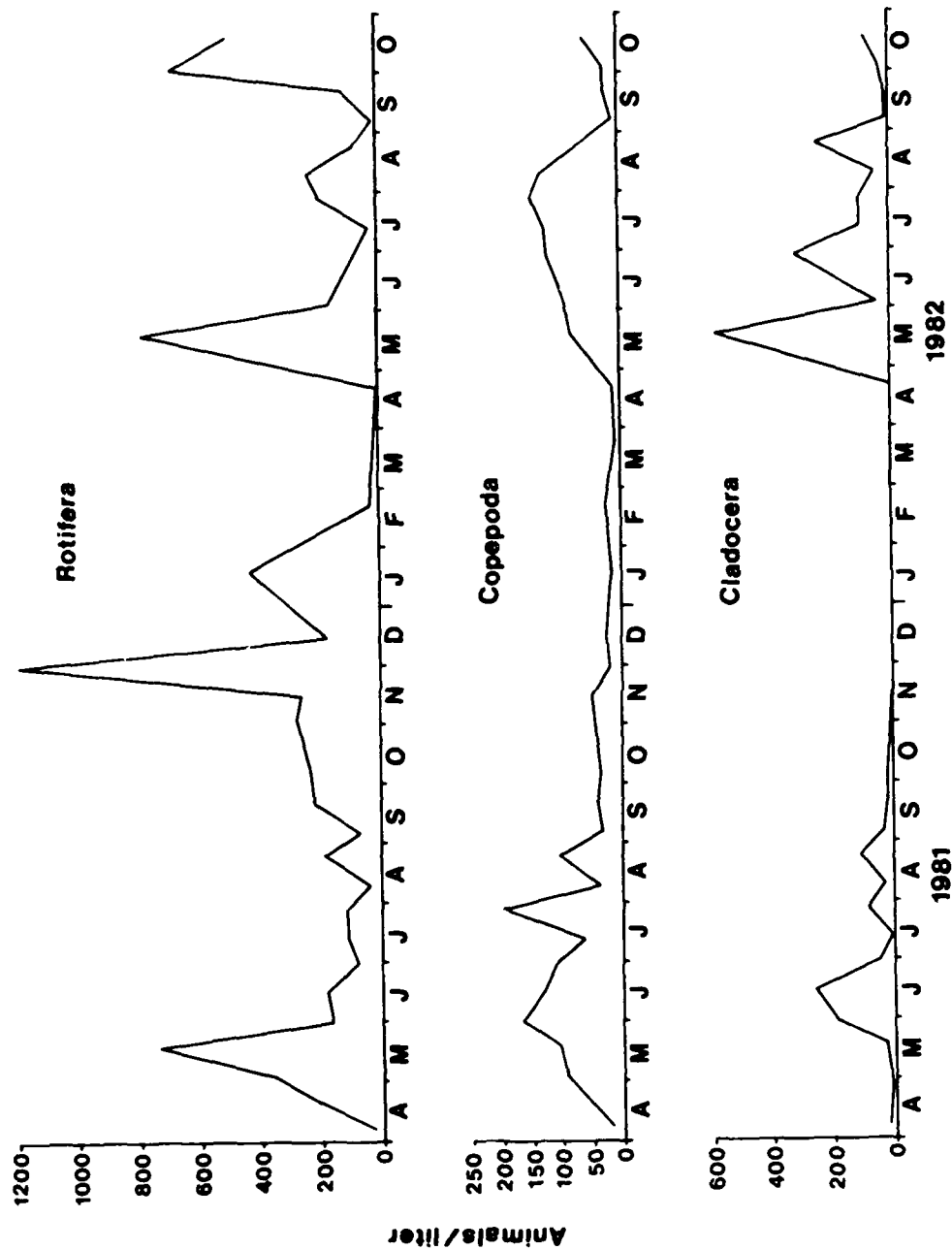


Figure 16. Seasonal trends of major zooplankton groups at station 50

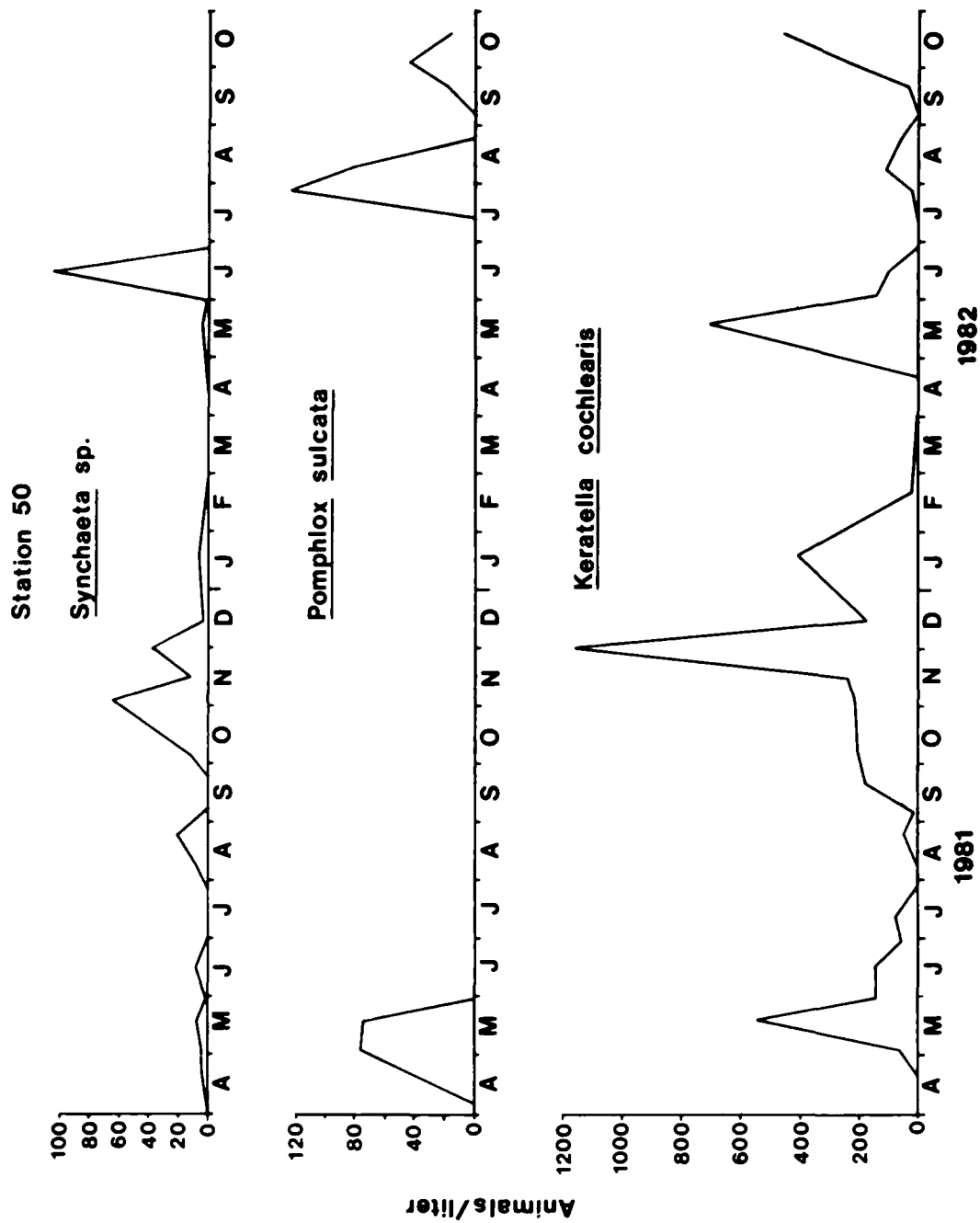


Figure 17. Seasonal changes of some dominant rotifers in the littoral zone. Note the differences in the vertical scales

Station 50

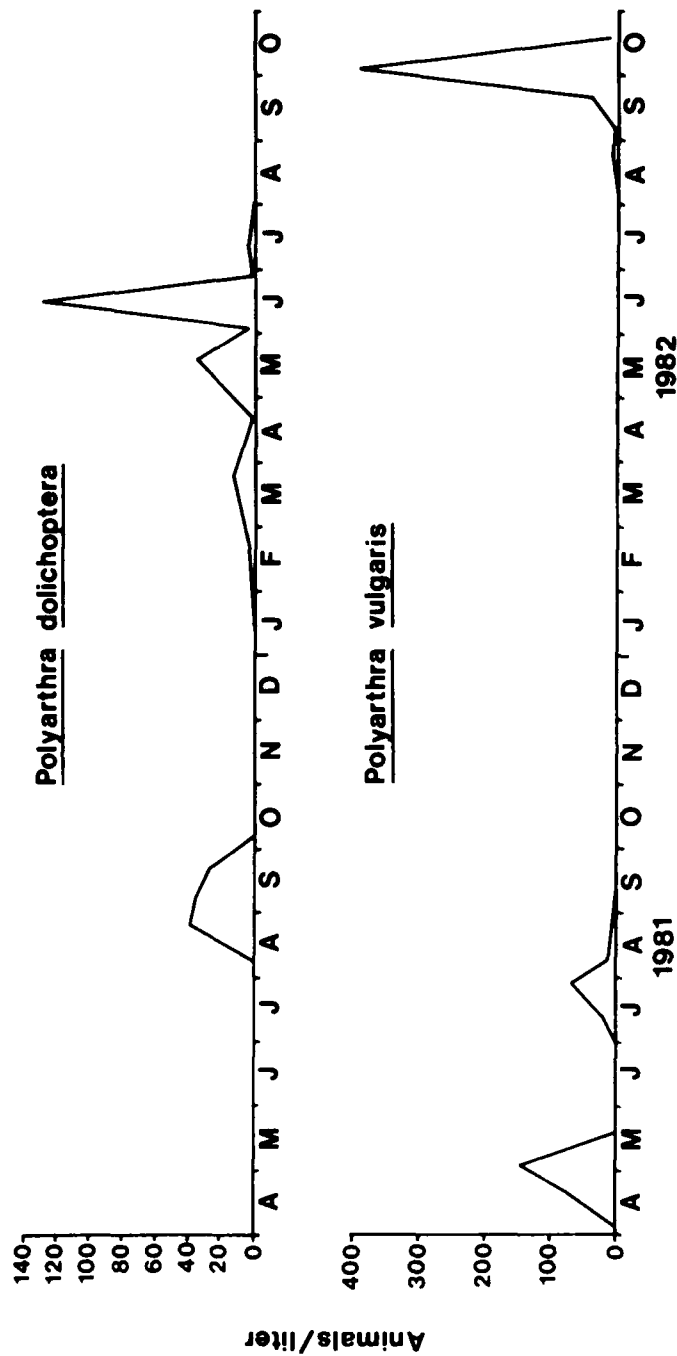


Figure 18. Seasonal changes of some dominant rotifers in the littoral zone. Note the differences in the vertical scales

Station 50

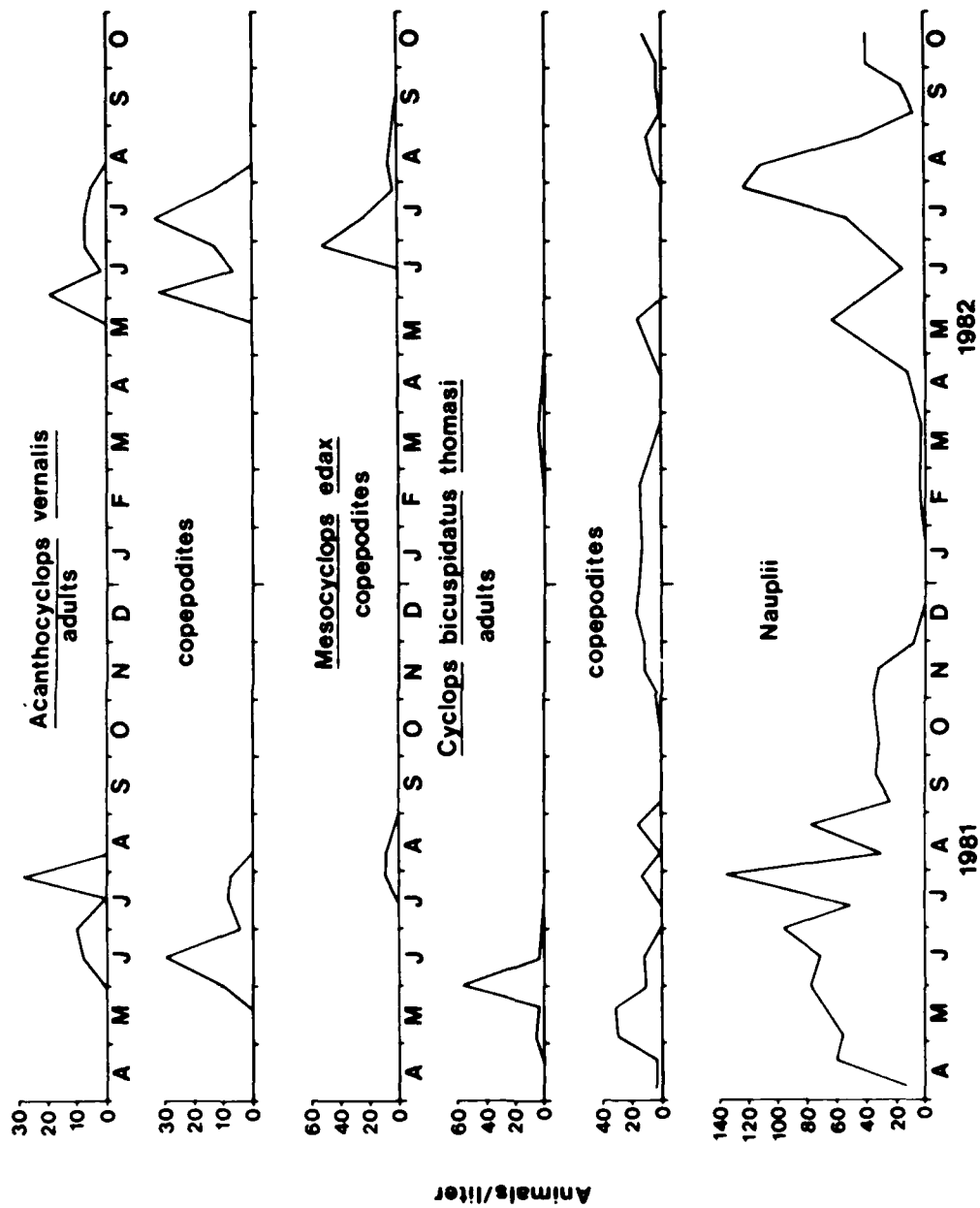


Figure 19. Seasonal trends of dominant copepods at station 50

nauplii were probably cyclopoids. Nauplii were more abundant in 1981 than 1982, but similar peaks occurred both years (Figure 19).

74. *Bosmina longirostris* exhibited a single peak in June 1981 (Figure 20). However, in 1982 it exhibited three peaks, reaching nearly 600 organisms/liter during the spring peak. While *Daphnia* spp. exhibited a peak in early summer 1981, they were present in low numbers in 1982. In contrast, *Ceriodaphnia lacustris* was uncommon in 1981 but exhibited a brief peak during the summer of 1982. *Chydorus sphaericus* was more common in 1981 than 1982. It peaked during the summer, reaching levels of 80 organisms/liter.

Discussion

75. Gliwicz (1969) found that the percentage composition of rotifers was higher in eutrophic lakes. Gannon and Stemberger (1978) also noted the same trend. Even though the physical environment differs between stations 20 and 50, rotifers still dominate the zooplankton community. Although station 50 was located in the littoral area with the attendant macrophytes, the rotifer community was similar. *Keratella cochlearis*, which is common in eutrophic lakes, was by far the most common rotifer. Pourriot (1977) states that *Keratella* and *Polyarthra* feed primarily on cryptomonads and chrysomonads. He also noted that *Keratella* but not *Polyarthra* ingested detritus and its associated bacteria. Bogdan and Gilbert (1982) observed that *K. cochlearis* feed on a wide range of food particles. In contrast, *P. vulgaris* and *P. dolichoptera* only feed on flagellated cells. They attributed this to the functional morphologies of the corona and mastax. The corona of *Keratella* is designed to collect and process many particles at once and to efficiently handle a wide variety of sizes and shapes. On the other hand, the corona of *Polyarthra* is designed to capture individual cells. Thus, *Keratella* is able to be more competitive and therefore dominate the rotifer community. While station 50 had more species than station 20, the additional species were present only in low numbers.

Station 50

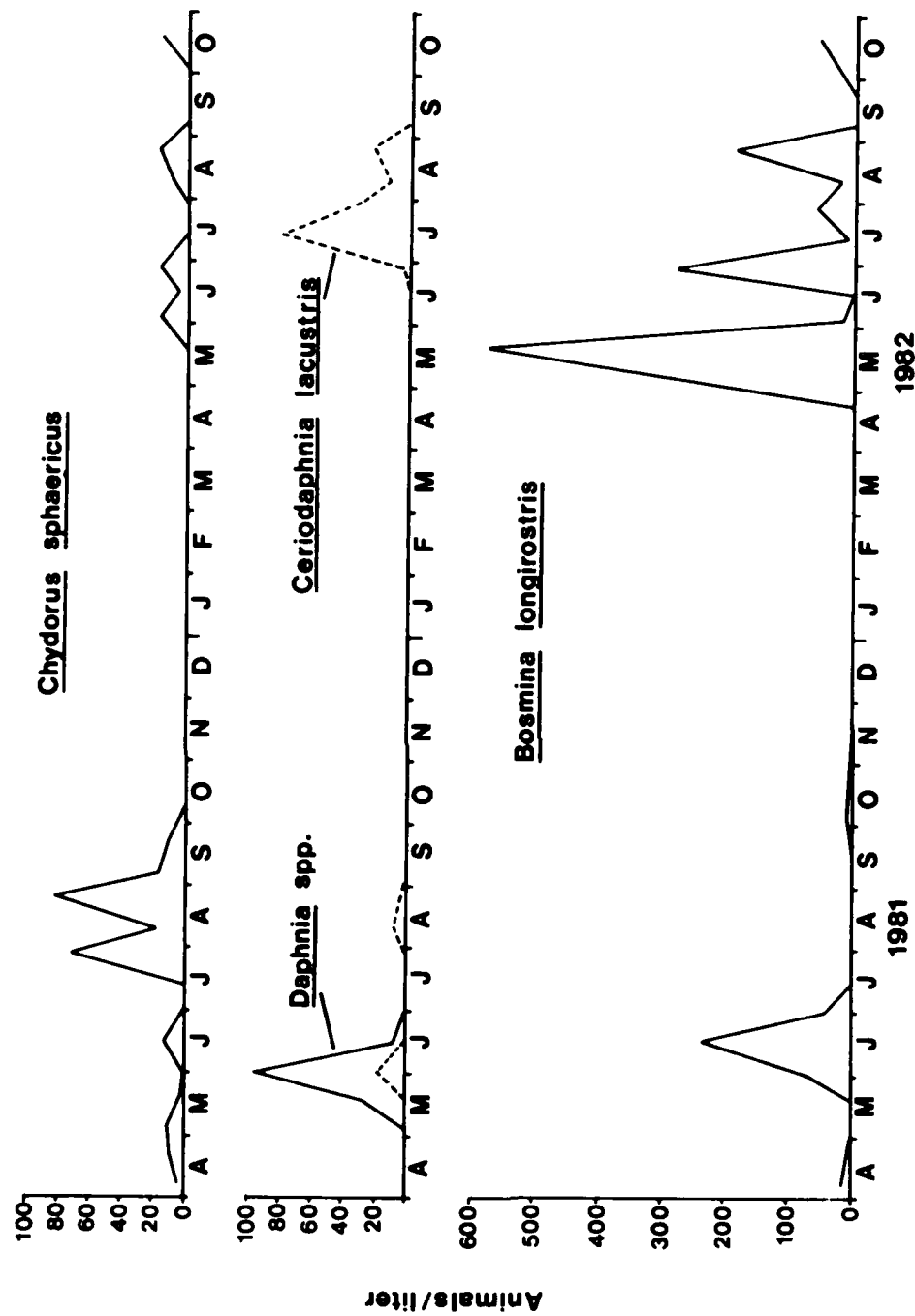


Figure 20. Seasonal changes of the important cladocerans at station 50

These additional species are most frequently found in littoral environments.

76. There probably was little predation on the rotifer community. The predaceous rotifer *Asplanchna* was present for brief periods of time. Although the cyclopoid copepods present at both stations are capable of feeding on a variety of rotifers, they are not capable of feeding on *K. cochlearis* (McQueen 1969, Williamson and Gilbert 1980). This may explain why this was the dominant rotifer in Eau Galle Lake. The *Asplanchna* population may be low because *Mesocyclops edax* readily preys upon this rotifer (Williamson and Gilbert 1980). In fact, *Asplanchna* generally was present only when *M. edax* was not.

77. While the rotifer community was regulated by food supply with little influence from predation, this was not the case with the crustaceans. Although diet certainly was an important factor in community composition, predation from both vertebrates and invertebrates was important in structuring the community. Hall, Cooper, and Werner (1970) and Lynch (1979) noted that fish predation can exert an overwhelming effect on the species composition of a zooplankton community. Numerous studies have documented the effects of size-selective predation, both in terms of the size-frequency distributions of a given food item and subsequent effects on community structure (Brooks and Dodson 1965, Galbraith 1967, Brooks 1968, Kitchell and Kitchell 1980). In such studies, a consistent pattern emerges in which the primary food item (usually large *Daphnia*) undergoes length reduction, followed by its eradication (in some cases) and subsequent replacement by less preferred or smaller zooplankters. The replacement species is usually a smaller cladoceran such as *Bosmina* or *Ceriodaphnia*.

78. At station 50 most of the cladocerans were *Bosmina longirostris*. This cladoceran is small and resistant to fish predation. Heavy fish predation would be expected in the littoral areas because fish such as bluegills are most abundant here. In contrast, at station 20 fish predation appeared to be less because the cladoceran community was dominated by *Daphnia* spp. The higher numbers of cladocerans at station 50 reflect the smaller body size of *Bosmina*. Larger sized

cladocerans (e.g., *Daphnia*) generally have a higher filtering rate than small-sized cladocerans (e.g., *Bosmina*) (Haney 1973). Taking differences in body size into account, the cladocerans at both stations probably consume similar amounts of food.

79. In early summer 1982 *B. longirostris* reached its greatest abundance at station 20, but was quickly replaced by *D. galeata mendotae*. DeMott and Kerfoot (1982) found that *Bosmina* and *Daphnia* could coexist under equilibrium conditions. While *Daphnia* could severely depress *Bosmina*, it could not eliminate it. Although they both use the same food resource, *Bosmina* is able to more efficiently capture highly edible flagellates by its feeding behavior. This ability may have depressed the rotifer community, due to the fact that both *Keratella* and *Polyarthra* feed on these flagellates. Since flagellates were still present in June, another factor may have helped eliminate the *Bosmina* population. Kerfoot (1977) noted that *Cyclops bicuspidatus thomasi* readily preyed upon *Bosmina*. While *Bosmina* can combat this by increasing the length of its antennules and micro spines, the cost is a reduction of egg production. *Bosmina* at station 20 did indeed have long antennules at this time. This morphological change in combination with competition from *D. galeata mendotae* probably caused the demise of the *Bosmina* population.

80. The appearance of *Diaphanosoma leuchtenbergianum* at station 20 occurs late in the summer when blue-green algae are an important part of the phytoplankton. Gliwicz (1969) found that *Diaphanosoma* did best when blue-greens were common because their filtering apparatuses were clogged less than other Cladocera such as *Daphnia* and *Bosmina*.

81. At station 50 *Ceriodaphnia lacustris* was infrequent in 1981, but there was a sizable population in July 1982. While it is unclear why it was more common in 1982, the presence of cyclopoid predators may have contributed. Jamieson (1980) has shown that *Mesocyclops* readily preys on *Ceriodaphnia* and prefers this over other copepodites.

82. *Chydorus sphaericus* is often found in the littoral zone of lakes. Individuals generally move about by crawling on attached substrates such as macrophytes. The predominant food source is epiphytes

which are attached to the macrophytes, although they also are known to filter feed (Fryer 1968, Gliwicz 1977). As with *L. leuchtenbergianus*, *C. sphaericus* is more competitive when blue-green algae are present (Gliwicz 1977). Because of its hard exoskeleton and distinctly rounded valves, it is not readily preyed upon by cyclopoids (Jamieson 1980). The reduction in the *C. sphaericus* population in 1982 was probably a reflection of the less abundant macrophytes.*

83. It is difficult to determine the reasons for seasonal fluctuations of cyclopoid copepods. The nauplii are herbivorous, feeding on small-sized algae and detritus, while the copepodites feed on phytoplankton at times but can also be carnivorous. McQueen (1969) and Anderson (1970) found that *Cyclops bicuspidatus thomasi* late stages of copepodites and adults fed on nauplii and young copepodites, especially diaptomids. *Acanthocyclops vernalis* is known to prey on small cladocerans such as *Bosmina* (Kerfoot 1977) while *Mesocyclops edax* feeds on *Ceriodaphnia* (Brandl and Fernando 1975).

84. The low numbers of diaptomids may partially be the result of the large numbers of predatory copepods found in Eau Galle Lake. Anderson (1980) found that diaptomids were uncommon in western Canadian lakes when there was a large population of cyclopoid copepods. *A. vernalis* is usually more carnivorous than the other two species of cyclopoids found in Eau Galle Lake. Since it is present at station 50 but not at station 20, the increased predation may exclude diaptomids from this area. The low populations of diaptomids in Eau Galle Lake were also due to its eutrophic nature. They usually are more common in lakes of lower trophic levels (McNaught 1975). It is common in a lake that becomes more eutrophic that the importance of the herbivorous diaptomids declines in favor of cyclopoids and cladocerans as well as rotifers (Ravera 1981).

* Personal Communication, 1986, Gerald J. Filbin, Environmental Laboratory, US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

85. Cyclopoid copepods seem to do better in eutrophic waters because they can change their source of food and they have a different mode of feeding than diaptomids. Cyclopoids feed by grasping their food and thus can be selective, while diaptomids are filter feeders. The cyclopoids in Eau Galle Lake probably are both herbivores and carnivores. Prey is generally limited during the ice-free period to nauplii, and at station 20 to *Liaphanosoma* and at station 50 to *Bosmina* and occasionally *Ceriodaphnia*. Heavy predation on nauplii probably does not occur as this would be detrimental to the cyclopoid community. In addition, Jamieson (1980) found that cyclopoid adults and late copepodites had difficulty capturing cyclopoid nauplii because of their swimming motions. Although the rotifer community was large at times, it was dominated by *Keratella cochlearis*. This rotifer, because of its hard exoskeleton, cannot be eaten by cyclopoids (McQueen 1969, Williamson and Gilbert 1980).

86. Predation by *C. b. thomasi* copepodites may have contributed to the *Bosmina* decline at station 20 in June 1982. *A. vermalis* may have had a strong effect in causing the decline in *Bosmina* at station 50 in May 1982. In contrast, *M. edax* does not seem to have affected the *Bosmina* population. Both *M. edax* and *Bosmina* increase at the same time. However, soon after *A. vermalis* increases, there is a resultant decrease in the *Bosmina* population.

Summary

87. The rotifera were the dominant zooplankton group at both stations with *Keratella cochlearis* being the most common rotifer. Since *K. cochlearis* is not affected by the invertebrate predators found in the lake, its population is regulated by food supply and at station 50 by competition with *Bosmina*.

88. Predation from both vertebrates and invertebrates had a profound effect upon the cladoceran community structure. Fish predation in the littoral area reduced the large cladocerans such as *Daphnia*, resulting in the community being dominated by *Bosmina* and *Chydorus*. The

predaceous cyclopoid *Acanthocyclops vernalis* appears to somewhat suppress the *Bosmina* population during the summer. In the pelagial area, fish predation is reduced, resulting in the cladoceran community being dominated by *Daphnia*.

89. The cyclopoids dominated the copepod community at both stations. *Cyclops bicuspidatus thomasi* dominated during fall, winter, and spring. During the summer, *A. vernalis* was dominant in the littoral area and *Mesocyclops edax* dominated in the pelagial zone. Calanoids were present in significant numbers in the lake only during late summer and only in the pelagial zone.

References

- Anderson, R. S. 1970. Predator-prey relationships and predation rates for crustacean zooplankters from some lakes in western Canada. Can. J. Zool. 48:1229-1240.
- _____. 1980. Relationships between trout and invertebrate species as predators and the structure of the crustacean and rotiferan plankton in mountain lakes. Amer. Soc. Limnol. Oceanogr. Spec. Symp. 3:635-641. Univ. Press New England.
- Bogdan, K. G., and J. J. Gilbert. 1982. Seasonal patterns of feeding by natural populations of *Keratella*, *Polyarthra*, and *Bosmina*: clearance rates, selectivities, and contributions to community grazing. Limnol. Oceanogr. 27:918-934.
- Brandl, Z., and C. H. Fernando. 1975. Food consumption and utilization in two freshwater cyclopoid copepods (*Mesocyclops edax* and *Cyclops vicinus*). Int. Rev. Gesamten. Hydrobiol. 60:471-494.
- Brooks, J. L. 1968. The effects of prey size selection by lake planktivores. Syst. Zool. 17:272-291.
- Brooks, J. L., and S. I. Dodson. 1965. Predation, body size, and composition of plankton. Science 150:28-35.
- DeMott, W. R., and W. C. Kerfoot. 1982. Competition among cladocerans: nature of the interaction between *Bosmina* and *Daphnia*. Ecology 63:1949-1968.
- Fryer, G. 1968. Evolution and adaptive radiation in the Chydoridae (Crustacea: Cladocera): a study in comparative functional morphology and ecology. Phil. Trans. Royal Soc. London Ser. B. 269:137-274.
- Galbraith, M. G., Jr. 1967. Size-selective predation on *Daphnia* by rainbow trout and yellow perch. Trans. Amer. Fish. Soc. 96:1-10.

- Gannon, J. E., and R. S. Stemberger. 1978. Zooplankton (especially crustaceans and rotifers) as indicators of water quality. *Trans. Amer. Micros. Soc.* 97:16-35.
- Gliwicz, Z. M. 1969. Studies on the feeding of pelagic zooplankton in lakes with varying trophy. *Ekol. Pol. A.* 17:665-708.
- Hall, D. J., W. E. Cooper, and E. E. Werner. 1970. An experimental approach to the population dynamics and structure of freshwater animal communities. *Limnol. Oceanogr.* 15:839-928.
- Haney, J. F. 1973. An in situ examination of the grazing activities of natural zooplankton communities. *Arch. Hydrobiol.* 72:87-132.
- Jamieson, C. D. 1980. The predatory feeding of copepodid stages III to adult *Mesocyclops leuckarti* (Claus). *Amer. Soc. Limnol. Oceanogr. Spec. Symp.* 3:518-537. Univ. Press New England.
- Kerfoot, W. C. 1977. Implications of copepod predation. *Limnol. Oceanogr.* 22:316-325.
- Kitchell, J. A., and J. F. Kitchell. 1980. Size-selective predation, light transmission, and oxygen stratification: evidence from the recent sediments of manipulated lakes. *Limnol. Oceanogr.* 25:389-402.
- Lynch, M. 1979. Predation, competition, and zooplankton community structure: an experimental study. *Limnol. Oceanogr.* 24:253-272.
- McNaught, D. C. 1975. A hypothesis to explain the succession from calanoids to cladocerans during eutrophication. *Verg. Internat. Verein. Limnol.* 19:724-731.
- McQueen, D. J. 1969. Reduction of zooplankton standing stocks by predaceous *Cyclops bicuspidatus thomasi* in Marion Lake, British Columbia. *J. Fish. Res. Bd. Can.* 26:1605-1618.
- Pourriot, R. 1977. Food and feeding habits of Rotifera. *Arch. Hydrobiol. Beih. Ergeb. Limnol.* 8:243-260.
- Ravera, O. 1981. The influence of nutrient enrichment on freshwater zooplankton. Pages 210-217 in *Restoration of lakes and inland waters*. EPA 440/5-81-010. Water Reg. and Stds. US Environ. Prot. Agency, Washington, DC.
- Williamson, C. E., and J. J. Gilbert. 1980. Variation among zooplankton predators: the potential of *Asplanchna*, *Mesocyclops*, and *Cyclops* to attack, capture, and eat various rotifer prey. *Amer. Soc. Limnol. Oceanogr. Spec. Symp.* 3:509-517. Univ. Press New England.

PART V: GROWTH AND NUTRITION OF SUBMERSED AQUATIC MACROPHYTES*

Introduction

90. In small lakes where the littoral zone occupies a relatively large surface area, macrophyte production may comprise a major portion of ecosystem primary production (Wetzel and Hough 1973; Rich, Wetzel, and Thuy 1971). In most macrophyte species the principal source of mineral nutrients, particularly nitrogen and phosphorus, appears to be the sediment (Hill 1979; Patterson and Brown 1979; Carignan and Kalff 1980; Barko and Smart 1980, 1981). However, in nonrooted or weakly rooted species, uptake of mineral nutrients may occur primarily from the water. For example, in *Ceratophyllum demersum* L., the mass and adsorptive function of the below-ground organs appear to be minimal (Arber 1920, Best 1979). This and functionally similar species may at times compete with phytoplankton for dissolved nutrients in the open water.

91. Nutrients incorporated into macrophyte tissues during growth may be liberated to the water column during senescence, thus providing a source for phytoplankton utilization (Howard-Williams and Howard-Williams 1978, Howard-Williams and Davies 1979, Landers 1979, Carpenter 1980). In many lakes, macrophyte communities provide an important source of nutrients to the water, thereby influencing water quality (Landers 1982).

92. This study was designed to measure the annual growth and nutrient dynamics of the macrophyte community in a small temperate impoundment, Eau Galle Lake, Wisconsin. The lake, constructed in 1967, is a hardwater basin (average surface alkalinity = 3.5 meq/l) with a surface area of approximately 0.6 km² and a mean depth of 3.0 m. The littoral zone, as defined by the summer presence of aquatic macrophytes, is about 17 percent of the total surface area. The depth of the lake is controlled primarily by an overflow structure.

* Part V was written by Gerald J. Filbin and John W. Barko.

Materials and Methods

93. Submersed macrophyte growth in Eau Galle Lake was examined between April and November 1981. Four replicate transects were established perpendicularly outward from the shore along a distance corresponding to limits of the littoral zone. Based upon location and basin morphometry, these areas were representative of the littoral region as a whole.

94. Annual macrophyte growth was estimated by quantitative biomass sampling at four to seven randomly selected locations per transect every 3 to 4 weeks. Macrophyte harvesting procedures were modified from Westlake (1974), incorporating the use of a lucite enclosure to delimit sampling areas of 0.5 m^2 to a depth of 1.2 m. Macrophytes within the areal confines of the enclosure were harvested to the sediment surface using a long-handled whip cutter and a rake.

95. Seasonal variations in six different components of macrophyte standing crop were evaluated (Figure 21). Harvested samples were rinsed lightly with tap water to remove mud and sand. Twigs and branches of obvious terrestrial origin were removed by hand. Total standing crop (i.e., total aboveground plant mass within each sample) was separated by individual species, then (by species) into senescent tissue mass and biomass prior to drying. A small fragment (1 to 2 g fresh weight) of biomass from major species was placed into a bottle containing a measured volume of potassium bicarbonate buffer (200 mg/l) for removal of epiphytes. In the laboratory, epiphytes were "washed" from macrophyte tissues by inverting the bottle 50 times. Tissues were removed and rewashed in fresh buffer two additional times. The triple washes for each plant sample were pooled and the total volume measured. A subsample of the wash was filtered through a preweighed $0.8\text{-}\mu$ glass fiber filter. Filters were dried at 70° C for 24 hr and reweighed. Tightly attached epiphytes were not examined in this study.

96. Each of the plant fractions were oven-dried (70° C) to constant weight. Fractions were then recombined and ground in a Wiley Mill equipped with a 40-mesh screen in preparation for digestion and tissue

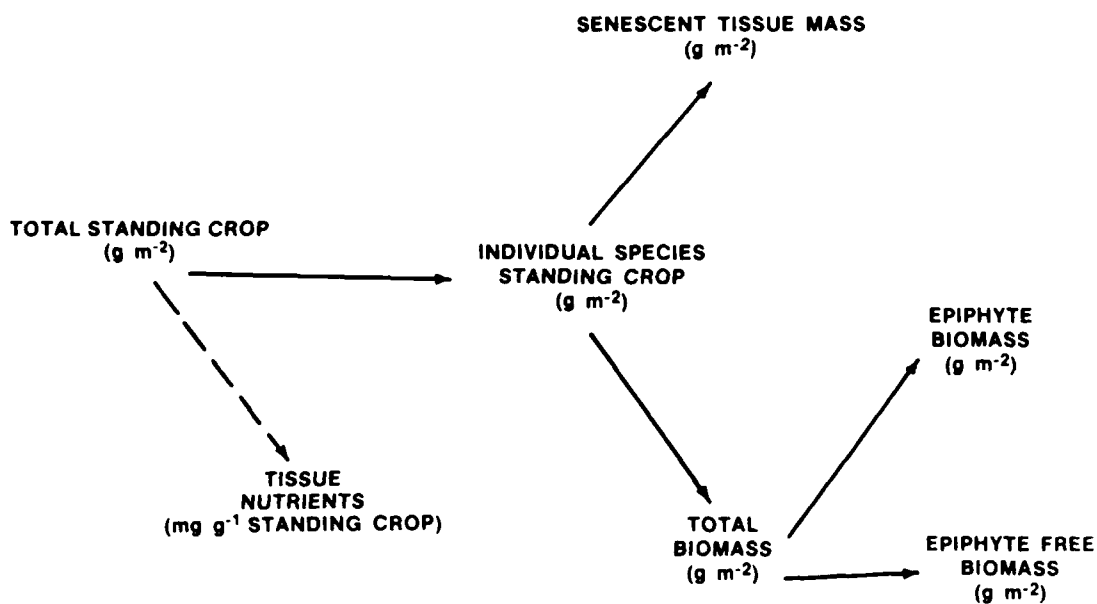


Figure 21. Sample components of total macrophyte standing crop measured during the 1981 growing season

nutrient analysis. Plant materials were digested using sulfuric acid and hydrogen peroxide (Allen et al. 1974). Phosphorus and nitrogen were analyzed colorimetrically using a Technicon Autoanalyzer. Potassium was analyzed by atomic absorption photometry. The accuracy of these analyses was routinely checked by the inclusion of National Bureau of Standards reference tissues in analytical sample sets. Nutritional data were obtained for only three of the four transects sampled, since one transect routinely yielded inadequate biomass for tissue nutrient analysis with sufficient replication.

Results

97. The submersed macrophyte community was dominated by *Ceratophyllum demersum* L. and *Potamogeton pectinatus* L., which respectively contributed 58 and 36 percent of maximum standing crop in 1981. Seasonal variations in macrophyte standing crop and component fractions thereof primarily reflect the integrated growth cycle of these two species (Figure 22). Minor species included *P. americanus* C. and S., *P. foliosus* Raf., and *Najas flexilis* (Willd.) Rostk. and Schmidt.

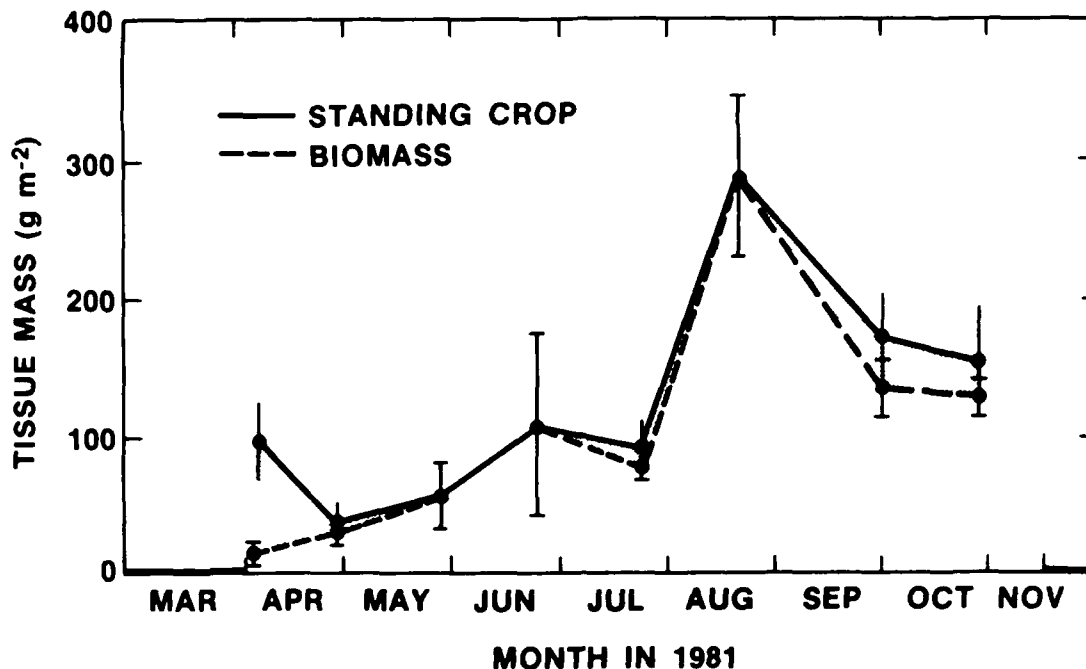


Figure 22. Macrophyte total standing crop and biomass in Eau Galle Lake during the 1981 growing season. Shaded area represents senescent tissue mass. Values are means \pm standard error. Thick regions of the abscissa indicate periods of ice cover

98. Following ice-out in late March, standing crop decreased to a seasonal minimum of ca. 12 g/m^2 in late April. Subsequent growth increased standing crop to 70 g/m^2 by late May and to ca. 300 g/m^2 by mid-August. Following this seasonal maximum, standing crop progressively declined to ca. 175 g/m^2 at the onset of winter.

99. Two different periods of senescence (early spring and fall) are suggested by differences between standing crop and biomass in Figure 22 and by variations in the seasonal growth cycle of *C. demersum* and *P. pectinatus* in Figure 23. The *Potamogeton* species began the season with virtually no standing crop and senesced in the fall. In contrast, standing crop in *C. demersum* persisted into the winter with a relatively low portion of senescent tissue in the fall. Major senescence in *Ceratophyllum* occurred prior to spring thaw, although release of decaying detritus was probably delayed until the ice melt.

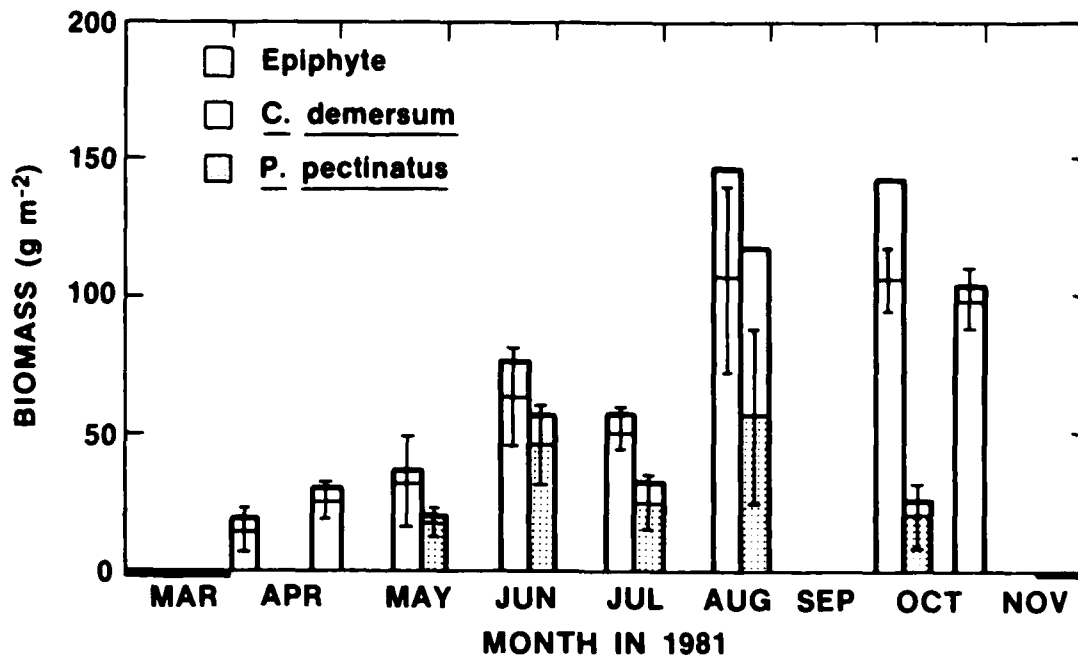


Figure 23. Epiphyte free biomass and epiphyte biomass in two principal macrophyte species in Eau Galle Lake during the 1981 growing season. Values are means \pm standard error.

Thick regions of the abscissa indicate periods of ice cover

100. Statistical variability associated with epiphyte biomass was at least as great as that associated with macrophyte biomass. Despite statistical overlap, however, epiphytes appeared to comprise a substantial portion of the littoral vegetative community, from 5 to 33 percent of total biomass (epiphytes plus macrophytes) in the two principal species (Figure 23). Epiphyte biomass associated with these species was proportionately similar throughout the growing season. During the late summer, epiphyte biomass was exceptionally high with filamentous algal mats positioned at or near the surface of the water. Principal algae associated with macrophytes were filamentous Chlorophyta and Cyanophyta, as well as numerous genera of pennate Bacillariophyta.

101. Macrophyte tissue concentrations of nitrogen (N), phosphorus (P), and potassium (K) (in milligrams per gram standing crop) varied to only a minor extent throughout the growing season. Thus, areal concentrations of tissue N, P, and K (grams per square meter) in the

macrophyte community varied directly with seasonal changes in standing crop (Figure 24). Seasonal average concentrations of tissue (standing crop) N, P, and K were 22.4, 3.2, and 18.7 mg/g, respectively, as estimated from slopes of regression lines in Figure 24. The overall N:P:K ratio is estimated at 7.0:1.0:5.4, similar to values provided by others for natural macrophyte communities (Hutchinson 1975, Carpenter and Adams 1977).

Discussion

102. It is unlikely that the growth of submersed macrophytes in Eau Galle Lake was limited by nutrients, since tissue N, P, and K concentrations were all well above the critical levels generally regarded as reflecting growth limitation (Gerloff and Krombholz 1966, Gerloff 1975). In contrast with the seasonal uniformity of macrophyte tissue nutrient concentrations here, seasonal variations in macrophyte tissue nutrients have been noted elsewhere (e.g., Kimball and Baker 1982). In our study, nutrient uptake was directly linked with macrophyte growth. However, tissue nutrient concentrations were possibly influenced by changes in epiphyte abundance and/or variations in proportions of dead to living tissue, since our estimates are based on total standing crop (not on macrophyte biomass per se).

103. Sculthorpe (1967) was doubtful that the stolons in *C. demersum* served any adsorptive function. However, Best (1977) presents evidence for this species indicating that P is absorbed from sediments while N uptake appears closely related to respective open-water concentrations. If nutrient uptake occurs predominantly from the aqueous phase in *C. demersum*, this species may compete for nutrients with the phytoplankton. Incorporation of N and P in the rooted submersed macrophytes of Eau Galle Lake likely occurred via nutrient mobilization from sediments (Barko and Smart 1980, 1981). Thus, competition for nutrients between *C. demersum* and more extensively rooted macrophyte species (*Potamogeton* species, in this study) may be reduced by specific distinctions in modes of nutrient uptake.

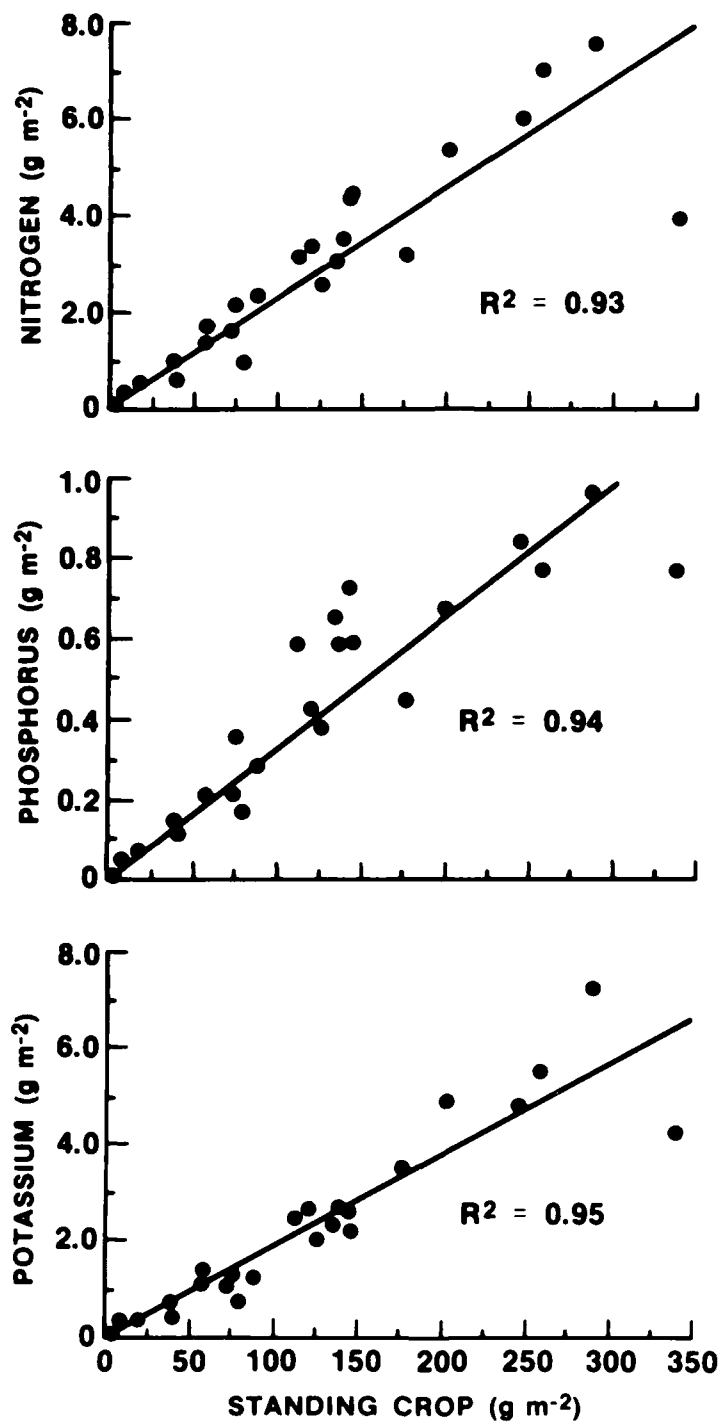


Figure 24. Littoral standing crop concentrations of N, P, and K in relation to macrophyte standing crop during the 1981 growing season. Values are means for three individual transects. Intercept in all regions is the origin

104. Epiphyte biomass in Eau Galle Lake was comparable to that determined in other eutrophic systems (Jones, Gurevitch, and Adams 1979; Borum and Wium-Andersen 1980; Gons 1982). Epiphyte biomass increased proportionately with standing crop. Accordingly, the increase in epiphyte mass per square meter during the late summer may reflect only the increase of macrophyte substrate available for growth. This, in combination with the proportionately similar epiphyte biomass with respect to host macrophyte species in Eau Galle Lake, supports the suggestion that macrophytes provide a neutral substrate for epiphytes (Cattaneo and Kolff 1979). Algal epiphytes and phytoplankton may be responsible for limiting the growth of macrophytes in Eau Galle Lake due to light attenuation, as suggested in the investigations of Phillips, Eminson, and Moss (1978) and Sand-Jensen and Sondergaard (1981).

105. The dominant macrophyte species in Eau Galle Lake, *Ceratophyllum demersum*, is well adapted to survival in cold climates (Holmquist 1971). However, during the winters of 1980-1981 and 1981-1982, qualitative sampling indicated considerable amounts of nonviable *C. demersum* frozen in the ice. Apparent winter kill of this species in Eau Galle Lake is consistent with the observations of Carr (1969) suggesting that turions of *C. demersum* are cold labile. Trapped in the ice, *C. demersum* provides a potential source of nutrients resulting from subsequent tissue decay during spring thaw.

106. Nutrients released by senescent *Ceratophyllum* may be flushed totally from the reservoir during the spring, as the hydraulic residence time is reduced to 4 days or less. Thus, the significance of nutrients contributed to the water column by this species likely depends upon the character of spring hydraulic conditions. Conversely, nutrients released from *Potamogeton* spp. in the fall during massive senescence and low-flow hydraulic conditions are likely retained by the system. Potential macrophyte P loading, estimated from standing crop concentrations, represented only about 0.2 percent of the annual P input from the watershed in 1981 (Kennedy 1986). This input appears to be a much smaller percentage of total P loading than for temperate lakes (e.g., Carpenter 1980). The apparently reduced significance of macrophyte nutrient

inputs to Eau Galle Lake, and perhaps reservoirs in general, is due primarily to the periodically riverine nature of impoundments (i.e., relatively short residence time) and the generally greater amplitude of loading peaks compared to natural lakes. The total macrophyte N, P, and K concentrations (on a lakewide basis) at the time of maximum biomass were estimated at 420, 62, and 425 kg, respectively.

Summary and Conclusions

107. The annual production of submersed macrophytes in Eau Galle Lake, Wisconsin, was measured during the 1981 growing season. While much of the total macrophyte standing crop accrued during the summer and senesced during the early fall, *Ceratophyllum demersum* L., the dominant species, senesced prior to the spring thaw.

108. Epiphyte biomass comprised a large portion of macrophyte biomass, ranging from 5 to 33 percent of total biomass for the two major macrophyte species. Changes in epiphyte biomass were related to changes in macrophyte standing crop. Macrophytes appeared to provide only a neutral substrate for epiphyte development.

109. Littoral concentrations of macrophyte tissue nutrients (N, P, and K) changed primarily as a function of change in standing crop rather than changes in tissue concentration. Release of tissue nutrients associated with macrophyte decay potentially occurred as two separate annual events, during the spring and fall. Despite the large littoral surface area, the significance of the macrophyte community as a source of nutrients is likely small, because of the periodically riverine nature of Eau Galle Lake.

References

- Allen, S. E., H. M. Grimshaw, J. A. Parkinson, and C. Quarmby. 1974. Chemical analysis of ecological materials. John Wiley and Sons Inc., New York.
- Arber, A. 1920. Water plants, a study of aquatic angiosperms. London. 436 pp.

- Barko, J. W., and R. M. Smart. 1980. Mobilization of sediment phosphorus by submersed freshwater macrophytes. *Freshwat. Biol.* 10:229-238.
- _____. 1981. Sediment based nutrition of submersed macrophytes. *Aquat. Botany* 10:339-352.
- Best, P. H. 1977. Seasonal changes in mineral and organic components of *Ceratophyllum demersum* and *Elodea canadensis*. *Aquat. Botany* 3:337-348.
- _____. 1979. Ecophysiological studies on growth and development of the aquatic macrophyte, *Ceratophyllum demersum* L. Doctoral Dissertation. Limnological Inst. Nieuwersluis, The Netherlands. 104 pp.
- Borum, J., and S. Wiium-Andersen. 1980. Biomass and production of epiphytes on eelgrass (*Zostera marina* L.) in the Oresund, Denmark. *Ophelia*. Suppl. 1:57-64.
- Carignan, R., and J. Kalff. 1980. Phosphorus sources for aquatic weeds: water on sediments. *Science* 207:987-989.
- Carpenter, S. R. 1980. Enrichment of Lake Wingra, Wisconsin, by submersed macrophyte decay. *Ecology* 61:1145-1155.
- Carpenter, S. R., and M. S. Adams. 1977. The macrophyte tissue nutrient pool of a hardwater eutrophic lake: implications for harvesting. *Aquat. Bot.* 3:239-255.
- Carr, J. L. 1969. The primary productivity and physiology of *Ceratophyllum demersum*; II. Micro primary productivity, pH, and the P-R ratio. *Aust. J. Mar. Freshwat. Res.* 20:127-142.
- Cattaneo, A., and J. Kalff. 1979. Primary production of algae growing on natural and artificial aquatic plants: A study of interactions between epiphytes and their substrate. *Limnol. and Oceanogr.* 24:1031-1037.
- Gerloff, G. C. 1975. Nutritional ecology of nuisance aquatic plants. National Environmental Research Center, Office of Research and Development, US Environmental Protection Agency, Corvallis, Oreg. 78 pp.
- Gerloff, G. C., and P. H. Krombholz. 1966. Tissue analysis as a measure of nutrient availability for the growth of aquatic plants. *Limnol. and Oceanogr.* 11:529-537.
- Gons, H. 1982. Structural and functional characteristics of epiphyton and epipelon in relation to their distribution in Lake Vechten. *Hydrobiologia* 95:79-114.
- Hill, B. H. 1979. Uptake and release of nutrients by aquatic macrophytes. *Aquat. Botany* 7:87-93.
- Holmquist, C. 1971. Northerly localities for three aquatic plants, *Potamogeton zosterifolius* L., *Ceratophyllum demersum* L., and *Myriophyllum spicatum* L. *Bot. Notiser, Lund.* 124:335-342.

- Howard-Williams, C., and B. R. Davies. 1979. The rates of dry matter and nutrient loss from decomposing *Potamogeton pectinatus* in a brackish south-temperate coastal lake. *Freshwat. Biol.* 9:13-21.
- Howard-Williams, C., and W. Howard-Williams. 1978. Nutrient leaching from the swamp vegetation of Lake Chilwa, a shallow African Lake. *Aquat. Botany* 4:257-267.
- Hutchinson, G. E. 1975. A treatise on limnology; III. Limnological Botany. John Wiley and Sons, New York.
- Jones, R. C., A. Gurevitch, and M. S. Adams. 1979. Significance of the epiphyte component of the littoral to biomass and phosphorus removal by harvesting. In J. B. Breck, R. T. Prentki, and O. L. Louks, eds. *Aquatic plants, lake management, and ecosystem consequences of lake harvesting*. Proceedings of conference at Madison, Wisconsin, February 14-16, 1979. University of Wisconsin, Madison, Wis.
- Kennedy, R. H. 1986. Material loadings to Eau Galle Lake. In R. H. Kennedy and R. C. Gunkel, Jr., eds. *Limnological studies at Eau Galle Lake, Wisconsin; Report 2, Special studies and summary*. Technical Report E-85-2. US Army Engineer Waterways Experiment Station, Vicksburg, Miss.
- Kimball, K. D., and A. L. Baker. 1982. Variations in mineral content of *Myriophyllum heterophyllum* Michx. related to site and season. *Aquat. Bot.* 14:139-149.
- Landers, D. H. 1979. Nutrient release from senescing milfoil and phytoplankton response. In J. B. Breck, R. T. Prentki, and O. L. Louks, eds. *Aquatic plants, lake management, and ecosystem consequences of lake harvesting*. Proceedings of conference at Madison, Wisconsin, February 14-16, 1979. University of Wisconsin, Madison, Wis.
- _____. 1982. Effects of naturally senescing aquatic macrophytes on nutrient chemistry and chlorophyll a of surrounding waters. *Limnol. Oceanogr.* 27:428-439.
- Patterson, K. J., and J. M. A. Brown. 1979. Growth and elemental composition of the aquatic macrophyte, *Sagittaria arifolia*, in response to water and substrate nutrients. *Progr. Wat. Tech.* 11:231-246.
- Phillips, G. L., D. Eminson, and B. Moss. 1978. A mechanism to account for macrophyte decline in progressively eutrophicated freshwaters. *Aquatic Bot.* 4:103-126.
- Rich, P. H., R. G. Wetzel, and N. V. Thuy. 1971. Distribution, production, and role of aquatic macrophytes in a southern Michigan marl lake. *Freshwater Biol.* 1:3-21.
- Sand-Jensen, K., and M. Sondergaard. 1981. Phytoplankton and epiphyte development and their shading effect on submerged macrophytes in lakes of different nutrient status. *Int. Revue Ges. Hydrobiol.* 66:529-552.
- Sculthorpe, C. D. 1967. The biology of aquatic vascular plants. St. Martin's Press, London. 610 pp.

Westlake, D. F. 1974. Macrophytes. In R. A. Vollenweider, ed. A manual on methods for measuring primary production in aquatic environments. I.B.P. Handbook No. 12. Blackwell Scientific Publications, Oxford.

Wetzel, R. G., and R. A. Hough. 1973. Productivity and role of aquatic macrophytes in lakes: an assessment. Pol. Archiv. Hydrobiol. 20:9-19.

PART VI: FACTORS DETERMINING THE DISTRIBUTION OF MACROINVERTEBRATES
ASSOCIATED WITH MACROPHYTES OF THE LITTORAL ZONE
IN EAU GAILLE LAKE*

Introduction

110. The littoral zone represents the interface between the terrestrial and open-water portions of a lake where components of the two systems merge and mix to form a potentially complex, dynamic, and productive ecotone (Wetzel 1975, Moore 1980). The chemical and physical characteristics of the littoral zone may be determined by factors that are either terrestrial (i.e., shading by lakeside trees, inputs of nutrients and organic and inorganic material) or limnetic (i.e., currents and wave action) in nature (Stolbunova and Stolbunov 1980). Littoral zones are also an important component of the lake ecosystem. This area may provide spawning sites, nursery areas, and foraging sites for fish. The often diverse and productive macroinvertebrate community associated with the littoral zone may link the production of the littoral zone to higher trophic levels (fish) in the lake. The heterotrophs in the littoral zone may recycle nutrients bound up in phytoplankton and macrophytes and facilitate the decomposition of allochthonous and autochthonous organic material.

111. From a practical standpoint, the littoral zone has the potential to impact recreation, water supply, navigation, and other beneficial uses of reservoir projects, particularly in shallow reservoirs. In the littoral zone, relatively high rates of respiration and photosynthesis concentrated in a relatively small volume of water can cause short- and long-term changes in water quality. Additionally, littoral zones may serve as "seed" areas that foster the development of nuisance accumulations of aquatic macrophytes that may impede recreation, navigation, and decrease the water quality and aesthetic value of the reservoir.

* Part VI was written by John Nestler.

112. The littoral zone is one of the least known components of the lake ecosystem despite the potential effect this area may have on conditions within the rest of the lake. The objective of this study is to identify biological, chemical, and physical factors that determine the number and distribution of members of one component of the littoral zone, the macroinvertebrate taxa in the littoral zone of a temperate lake. The resulting information will provide a better understanding of the littoral zone. In addition, the information generated by this study will foster a better conceptualization of the lake ecosystem and, in some instances, may provide information useful in predicting the consequences of reservoir operation on this group of organisms.

113. Several different broad classes of variables may influence the littoral zone benthic community. Food-related variables investigated in this study that may affect macroinvertebrate herbivores and detritivores include the mass of epiphytes growing on the macrophytes in the littoral zone, the mass of macrophytes, and the mass of senescent (decaying) macrophytes. Chlorophyll a concentration, an index of algal biomass, may be an important factor determining the distribution and abundance of filter-feeding invertebrates or invertebrates that feed on phytoplankton settling out on the bottom of the littoral zone. Some of the above variables may have secondary effects. Large amounts of senescent macrophytes may also impact water quality by serving as a substrate for aquatic heterotrophs. The macrophytes may also function as a substrate for some groups and particularly dense stands of macrophytes may provide refuge from vertebrate predators (fish).

114. Water quality variables investigated in this study that reflect substantial, general changes in the physicochemical conditions and may affect macroinvertebrates in the littoral zone include concentration of fecal coliform bacteria, specific conductivity, suspended solids, and total phosphorus. Concentration of fecal coliform bacteria provides an index of nutrient inputs caused by rainfall in the basin, particularly since the drainage area of each tributary includes a substantial amount of pastureland. Specific conductivity provides an insight into the concentration of nutrients available in the system.

Suspended solids concentration provides an estimate of particulate matter suspended in the water column. Total phosphorus values, in conjunction with chlorophyll a values, provide an estimate of the trophic status of the littoral zone.

Materials and Methods

115. Macroinvertebrates were collected at monthly intervals during the growing season (May through October) of 1982 at Eau Galle Lake. Macroinvertebrate samples used in the analyses were restricted to the north end of the lake in a large, shallow area typically dominated by aquatic macrophytes. The area sampled for macroinvertebrates corresponds to two macrophyte sampling transects. One sampling transect was located in a cove just northeast of the Eau Galle River inflow. The other sampling transect was located in a cove fed by Lousy Creek at the north end of Eau Galle Lake.

116. Macroinvertebrates were collected from a boat-mounted drop-chamber in conjunction with the sampling of aquatic macrophytes. After the macrophytes were removed from the sampling chamber, the chamber was agitated with a boat paddle to randomly distribute the macroinvertebrates. A sweep net was then drawn down the side of the chamber, across the open bottom of the chamber, and up the opposite side of the chamber. The contents of the sweep net were emptied into a museum jar filled with 10-percent buffered formalin. A total of four sweeps were made through the chamber at each sampling site. Macroinvertebrates that were later dislodged during the processing of the macrophyte samples were also saved in a separate vial. Sample vials were picked and the number of organisms per taxon was tallied. The organisms were identified to a taxonomic level sufficient to restrict the members of each category to a single general method of foraging.

117. Initially, attempts were made to estimate the number of organisms per taxon using removal sampling techniques (Carle and Strub 1978). Unfortunately, the variances associated with removal sampling techniques were unacceptably high, probably because the sweep net was

not drawn through the chamber in a systematic fashion during the course of this study. Therefore, each sweep was considered a replicate and the sweeps and rinse were pooled to obtain an index of relative abundance. This index was calculated as

$$\text{relative density} = (S1 + S2 + S3 + S4 + RN)/(2Z + 1)$$

where

S1 = the number per taxon in sweep one

S2 = the number per taxon in sweep two

S3 = the number per taxon in sweep three

S4 = the number per taxon in sweep four

RN = the number per taxon dislodged during processing of the
macrophyte samples

Z = the water depth inside the sampling chamber

The term in the denominator ($2Z + 1$) represents the distance that the net is drawn through the water within the chamber (the surface area covered by the chamber is 1.0×0.5 m).

118. Water quality and biological data used in the analysis were obtained from other members of this cooperative effort. The variables used in the analysis are listed in Table 6. Water quality data were collected at 2-week intervals at water quality stations 50 and 60 (Kennedy 1985). Water quality station 50 was associated with the macrophyte transect northeast of the Eau Galle River inflow, and water quality station 60 was associated with the macrophyte transect located in the cove of Lousy Creek. The water quality parameters and methods of collection, preparation, and analysis are presented in Kennedy (1985). Methods used to determine the taxonomic composition and biomass of macrophytes and the biomass of epiphytes are described by Filbin and Barko (1986). A mean was determined for each water quality variable using values from the immediately preceding two water quality sampling dates since water quality measurements were made at 2-week intervals and

Table 6
Variables Used in the Analysis

<u>Variable</u>	<u>Explanation</u>
LRELD	Base 10 logarithm of the relative density
LEPIW	Base 10 logarithm of epiphyte mass per square meter
LEPID	Base 10 logarithm of epiphyte mass per cubic meter
LMASS	Base 10 logarithm of macrophyte mass per square meter
LMASD	Base 10 logarithm of macrophyte mass per cubic meter
LRTM	Base 10 logarithm of senescent macrophyte biomass
M	Month in which the samples were collected
Z	Water depth within the sample chamber
CHLA	Water column chlorophyll <u>a</u> concentration, milligrams per liter
LFCOL	Base 10 logarithm of fecal coliform bacteria
SPCON	Specific conductivity
SS	Suspended solids
TP	Total phosphorus

macroinvertebrates were collected at monthly intervals. Month (M) was included as a variable to account for changes associated with naturally occurring seasonal life history changes.

119. All data manipulations and analyses were performed using Statistical Analysis System (SAS) Release 79.4 (Barr et al. 1979). The raw data were examined for skewness and kurtosis and, as a consequence, the dependent variable and some of the independent variables were transformed using a log transformation. Taxa of invertebrates associated with macrophytes had to occur at least 10 of a possible 20 times to be included in the regression analysis. In order to maximize the number of observations, the macrophyte biomass was pooled independently of species composition. Effects caused by pooling of the macrophyte biomass and other qualitative variables in the data set were assessed during examination of plots of the residuals versus predicted values.

Results

120. Variables that reflect conditions in the lake may often be highly correlated since the lake ecosystem is composed of many components that interact physically, chemically, and biologically. Correlation analysis indicated that a number of the variables investigated in this study were highly correlated (Table 7). Volumetric and areal measures of epiphyte and macrophyte density were highly correlated. However, none of these variables were dropped since they behaved differently in the analyses. The variable "month" (M) was significantly ($\alpha < 0.05$) related to all of the biological variables except fecal coliform bacteria concentration. This variable reflects the overall, progressive development of summer conditions from the first month sampled (April) to the last month sampled (October). Most of the other variables investigated in this study would also be sensitive to the same gradual, seasonal changes, and exhibit significant correlations to variable M. The mass and density of both epiphytes and macrophytes and the relative density of macroinvertebrates increased with the overall progressive development of summer conditions from April to October. Total phosphorus (TP) was significantly correlated with chlorophyll a (CHLA) and suspended solids (SS) since each of these variables reflects eutrophied conditions in the littoral zone. Increased numbers or volumes of algae contribute to SS, TP (since algae contain phosphorus), and CHLA (since chlorophyll a is an index of algal biomass).

121. The maximum R-squared improvement technique was used to develop the best regression models (Table 8) using a maximum of four of the independent variables listed in Table 6. The most appropriate model was selected based upon the R-squared statistic, assessment of the residual mean square with increasing number of variables, and Mallows Cp statistic (Draper and Smith 1981). In general, an independent variable had to be significant at the $\alpha = 0.05$ level to be included in the analysis. However, in two instances, slightly less significant variables were included because they resulted in a marked increase in the R-squared statistic. Note that, in general, the analyses appear quite

Table 7
Pearson Product-Moment Correlation Coefficients
of Independent Variables

	<u>LRELD</u>	<u>SPCON</u>	<u>TP</u>	<u>CHLA</u>	<u>SS</u>	<u>Z</u>	<u>LEPIW</u>	<u>LMASS</u>	<u>LMASD</u>	<u>LEPID</u>	<u>LTRM</u>	<u>IFCOL</u>
M	0.32	0.68	0.04	0.48	-0.11	0.04	0.68	0.64	0.56	0.62	0.54	-0.03
	*	*	ns	*	*	ns	*	*	*	*	*	ns
LRELD		0.10	0.16	0.30	0.05	-0.05	0.29	0.26	0.21	0.24	0.28	-0.05
		ns	*	*	ns	ns	*	*	*	*	*	ns
SPCON			-0.57	-0.04	-0.44	-0.24	0.32	0.21	0.15	0.26	0.37	0.29
			*	ns	*	*	*	*	*	*	*	*
TP				0.60	0.84	0.38	0.19	0.32	0.33	0.18	0.06	-0.03
				*	*	*	*	*	*	*	ns	ns
CHLA					0.30	-0.06	0.33	0.33	0.24	0.23	0.25	-0.13
					*	ns	*	*	*	*	*	*
SS						0.39	0.09	0.19	0.24	0.12	-0.15	0.26
						*	ns	*	*	*	*	*
Z							0.16	0.39	0.52	0.28	-0.17	-0.15
							*	*	*	*	*	*
LEPIW								0.69	0.63	0.98	0.21	0.26
								*	*	*	*	*
LMASS									0.98	0.66	0.16	0.16
									*	*	*	*
LMASD										0.63	0.07	0.13
										*	ns	*
LEPID											0.11	0.23
											*	*
LTRM												-0.22
												*

* Significant at $P < 0.05$.

ns Not significant ($P > 0.05$).

Table 8
Results of Multiple Regression Analysis

Taxon	R ²	Intercept	Regression Equation and Probabilities			
Oligochaeta	0.71	0.15	-1.15(Z) + 0.46(LEPIW) + 0.11(SS)	0.0160	0.0007	0.0003 /0.0001
Hydracarina	0.71	2.88	1.34(LEPIW) + -1.26(LEPID) + -0.01(SPCON)	0.0033	0.0174	0.0001 /0.0001
Chironominae	0.74	-0.26	0.46(LEPIW) + 0.06(SS) + 0.01(CHLA)	0.0011	0.0157	0.0534 /0.0001
Tanypodinae	0.60	0.04	0.96(LEPIW) + -0.89(LEPID) -0.21(LFCOL)	0.0083	0.0394	0.0170 /0.0016
Orthocladinae	0.47	-0.26	4.58(TP) + 0.25(LFCOL)	0.0232	0.0058	/0.0042
Ceratopogonidae	0.75	-0.73	-0.83(Z) + 0.23(M)	0.0146	0.0001	/0.0001
Ephemeroptera <i>Caenis</i>	0.46	0.10	0.01(CHLA) + 0.31(LRTM)	0.0019	0.0532	/0.0007
Coenagrionidae	0.66	0.74	-1.46(Z) + 0.89(LMASD) + -0.48(LFCOL)	0.0088	0.0001	0.0052 /0.0005
Trichoptera <i>Leptocerus</i>	0.36	0.67	0.44(LMASD) + -0.62(LFCOL)	0.0683	0.0157	/0.0237
Amphipoda <i>Hyallela</i>	0.68	-0.53	-0.73(Z) + 0.17(M)	0.0252	0.0001	/0.0001
Cladocera	0.72	0.55	0.494(LEPIW) + -0.10(SS) + 0.01(CHLA)	0.0015	0.0011	0.0105 /0.0001
Ephemeroptera <i>Baetis</i>	0.82	0.12	-0.37(Z) + 0.01(CHLA) + 0.14(LRTM)	0.0289	0.0001	0.0303 /0.0001
Pelecypoda	0.68	-0.07	0.004(CHLA) + 0.46(LRTM)	0.0278	0.0002	/0.0001
Dytiscidae	0.53	-0.02	0.27(LRTM)	0.0003		/0.0003
Pleidae	0.74	0.12	-0.02(SS) + 0.003(CHLA) + 0.19(LRTM)	0.0222	0.0016	0.0118 /0.0001

Note: Probabilities for each variable occur immediately under the regression equation and the probability level of the equation occurs to the right of the slash.

robust with the exception of the *Leptoclema* group which had an R squared of only 0.36 (Table 8). In most of the remaining cases the R-squared value was substantially above 0.50.

122. The general response of the macroinvertebrate community can be presented by listing the independent variables used in the analysis along with the number of significant, positive, and negative occurrences (Table 9). Several important factors become immediately obvious from an examination of Table 9. First, the abundance of all invertebrates in the littoral zone is determined primarily by CHLA, depth (Z), and weight of epiphytes (LEPIW). These variables account for almost one-half the significant occurrences of the independent variables. Also, the two variables describing the mass (LMASS) and density (LMASSD) of macrophytes had only 0 and 2 occurrences, respectively, in all of the equations.

Discussion

123. The distribution of major groups of macroinvertebrates within the littoral zone is based on only several of the variables investigated in this study (Table 9). Consequently, a discussion of these few variables may provide considerable information on factors that determine the abundance and distribution of the macroinvertebrates and also provide an insight into the dynamics of the littoral zone.

124. The abundance of potential food sources, as reflected by CHLA, mass of epiphytes (LEPID), and mass of senescent vegetation (LRTM), appears to be the most important general factor determining the structure of the littoral zone macroinvertebrate community. Both CHLA and LEPIW are important in determining the number of grazers, such as *Oligochaeta*, *Cladocera*, *Hydracarina*, *Baetis*, *Caenis*, and *Chironominae* (this tribe of the *Chironomidae* is composed of both predators and grazers; at least some of this group would be expected to be grazers), present in the samples. The importance of periphyton production to the abundance of chironomids (Mason and Bryant 1975, Dermott et al. 1977) and oligochaetes (Moore 1980, McElhone 1982) is well-documented for a number of systems. Cattaneo (1983) has even suggested that oligochaetes

Table 9
List of Independent Variables with Number of Significant
Positive and Negative Occurrences

<u>Independent Variable</u>	<u>Occurrences</u>		
	<u>Significant</u>	<u>Positive</u>	<u>Negative</u>
CHLA	6	6	-
Z	5	-	5
LEPIW	5	5	-
LRTM	5	5	-
SS	4	2	2
LFCOL	4	1	3
LMASD	2	2	-
LEPID	2	-	2
M	2	2	-
SPCON	1	-	1
TP	1	1	-
LMASS	-	-	-

and chironomids may control the species composition and seasonal abundance patterns of epiphytes. However, Moore (1980) did not observe a positive relationship between periphyton concentration and chironomid abundance in a shallow bay of Great Slave Lake.

125. The quantity of senescent macrophytes in each sample was also very important in determining the number of macroinvertebrates. However, this variable probably has a different effect on different taxa since two of the groups (Dytiscidae and Pleidae) are carnivores, whereas the rest are herbivores or detritivores (*Baetis*, *Caenis*, and *Pelecypoda*). As Street and Titmus (1982) have noted, coarse organic material in a lake may provide food, an appropriate substrate, or cover or a combination of the three depending upon the macroinvertebrate group in question. All of the above effects were probably observed in this study. The decaying macrophytes may depress dissolved oxygen levels to a degree that favors surface-breathing predators, such as Dytiscidae and Pleidae, over predators that obtain oxygen dissolved in the water column. Additionally, these two predators may be able to effectively

forage in the dense matrix of decaying vegetation that would ordinarily exclude predators such as fish. The ephemeropterans *Baetis* and *Caenis* were both generally found in areas with high concentrations of decaying organic matter. The reasons for this association are unclear. Several explanations may be plausible. The two genera may be attracted to the detritus for feeding, or the detritus may provide cover and refuge from fish. Alternatively, the presence of chlorophyll *a* as an important variable suggests that these two genera may be feeding on periphyton and sedimented phytoplankton whose growth would be enhanced by nutrients released from decaying macrophytes (Lander 1982). No general statements can be made about the pelecypods since they were not identified below class.

126. Depth was the next most important variable in explaining the abundance of macroinvertebrates. In all cases, the loadings for this variable were negative, indicating that some groups of macroinvertebrates were concentrated in the shallower portions of the littoral zone. Four of the five groups for which this variable was important, Oligochaeta, Ceratopogonidae, *Hyalella*, and *Caenis*, are generally associated with organically enriched silt and decaying vegetation common in very shallow water. Organic inputs to these areas originate from decaying emergent vegetation growing around the lake periphery and wind drift from the open portions of the lake. A further indication of eutrophied conditions in the shallower parts of the littoral zone is the presence of a dense community of duckweeds found in this area. An inverse relationship between depth and the relative density of several of the macroinvertebrate groups is important from a reservoir management standpoint because it suggests that summertime reservoir water-level fluctuations and drawdowns could have a substantial negative impact on the littoral zone macroinvertebrate community since many of these organisms may be stranded and desiccated. Benson and Hudson (1975) noted substantial increases in numbers of chironomids, *Hexagenia*, *Caenis*, oligochaetes, and ceratopogonids in Lake Francis Case when the extent of drawdown was reduced. The taxa at Francis Case Lake are very similar to the taxa collected in Eau Galle Lake, suggesting that summer drawdown may destroy

substantial numbers of the invertebrates which control the biomass of epiphytes in the littoral zone. Reflooding may then potentially produce a proliferation of epiphyte biomass.

127. The two variables that describe the quantity of living macrophytes (LMASS and LMASD) were not important in directly predicting the number of macroinvertebrates found in the littoral zone. Macrophytes certainly function as a substrate for epiphytes and provide detritus when they die, but when alive, apparently are not used directly as a food source by a significant portion of the macroinvertebrate community. Of the two groups that did react to this variable, one was a predator (Coenagrionidae) and the other a grazer (*Leptocerus*) generally associated with *Myriophyllum* (Ross 1944). The Coenagrionidae were probably using the vegetation for protection from fish and as a foraging ground since predation by fish is impeded by dense cover (Savino and Stein 1982). The trichopteran *Leptocerus* was also probably protected to some degree from fish predation by the dense cover. Note that in both cases the density of macrophytes and not the weight was the variable selected in the analysis.

128. The other independent variables were considered unimportant from a community standpoint either because they exhibited no consistent pattern of positive and negative effects or because they occurred too infrequently. The remaining independent variables can be best discussed within the context of their contribution to the abundance of a specific taxon (Table 8).

Oligochaetes

129. Feeding of oligochaetes on periphyton is well documented in the literature and is further substantiated in this investigation. Abundance of this taxon is greatest in shallow water containing substantial amounts of epiphytes. Additionally, the high loading (in a statistical sense) of suspended solids suggests that substrates composed of fine silt attract this group.

Hydracarina

130. The water mites are a diverse group with many very specific structural, life history, and behavioral adaptations that preclude

presentation of statements concerning their relationship to specific variables.

Chironominae

131. The abundance of this group (probably composed of both predators and grazers) in the Eau Galle Lake investigation generally follows the trends documented in the literature for the Chironomidae. The three variables in the model all represent or are indices of food availability.

Tanypodinae

132. This predatory tribe of the Chironomidae feeds primarily on other chironomids. The variable that explained most of the variance in abundance of the prey of this group (LEPIW) was also the most important variable for this group. Remarkably, the loading of density of epiphytes (LEPID) was negative even though the correlation coefficient between the two variables is 0.98. The negative loading of epiphyte density in conjunction with the negative loading of fecal coliform bacteria concentration suggests that very dense algal masses and associated high nutrient concentrations and poor water quality are detrimental to this taxon in Eau Galle Lake.

Orthocladiinae

133. The abundance of this predominantly herbivorous group is enhanced by eutrophied conditions in the littoral zone as indicated by the positive loadings of total phosphorus and fecal coliform bacteria concentration.

Ceratopogonidae

134. This family of dipterans is generally associated with organically enriched sediments in shallow or damp substrates. This association is reflected in the negative effects of depth. No other variable was significant except month (M) which, in this case, probably represents little more than the progression of summertime conditions in the littoral zone.

Ephemeroptera (*Caenis*, *Baetis*)

135. Variables explaining the abundance of these two ephemeropterans were discussed above. The only difference indicated by this

analysis is that *Baetis* is more restricted to shallow water. Baetids in streams often feed on periphyton on the upper sides of cobble (Elliott 1969). They probably occur no deeper than the photic zone.

Coenagrionidae

136. The abundance of this family of the Odonata is greatest in shallow water in dense beds of macrophytes, probably because these areas afford the greatest protection from fish (primarily centrarchids) which may prey heavily on dragonfly nymphs. The significance of the negative loading of fecal coliform bacteria is unknown.

Leptocerus (Trichoptera)

137. This small case-building caddisfly is generally associated with *Myriophyllum*. Since the independent variable selected in the regression is density of macrophytes and not mass, then the macrophytes are probably providing cover and substrate and not food. The high negative loading of fecal coliform bacteria also suggests that general water quality considerations may be important in determining the abundance of this genus.

Amphipoda

138. The factors that determine the abundance of this taxon are identical to the factors identified for the Ceratopogonidae.

Cladocera

139. These organisms were not keyed out below the family level and only the largest Cladocerans would be included in the sample because of the large sieve size used. However, the high R-squared value ($R^2 = 0.72$) observed for this group indicates that the results should be presented. The abundance of this taxon is predominantly determined by variables that reflect the availability of food (mass of epiphytes and chlorophyll a concentration).

Pelecypoda

140. Pelecypoda were most abundant in areas containing large amounts of senescent aquatic macrophytes. The reasons for this association are unclear.

Dytiscidae and Pleidae

141. These families were both associated with senescent aquatic vegetation. Since they are surface-breathing predators, they may be able to effectively forage in the matrix of decaying vegetation, or the decaying vegetation may cause localized water quality changes that exclude organisms, such as fish, that obtain oxygen from the water column.

Conclusions

142. Several important findings were made in this study.

- a. The experimental design of this study provided robust results. A similar approach may be very productive in further investigations of littoral zone macroinvertebrates.
- b. The abundances of major groups of macroinvertebrates were explained by a relatively small number of the variables examined, the most important of which were related to food (chlorophyll a, mass of epiphytes, and mass of senescent macrophytes) and depth.
- c. The inverse relationship between depth and some groups of invertebrates demonstrated in this and other studies suggests that these organisms may be very susceptible to even relatively small summertime drawdowns.

References

- Barr, A. J., J. H. Goodnight, J. P. Sall, W. H. Blair, and D. M. Chilko. 1979. SAS User's Guide. SAS Institute, Raleigh, N. C.
- Benson, N. G., and P. L. Hudson. 1975. Effects of a reduced fall draw-down on benthos abundance in Lake Francis Case. Trans. Am. Fish. Soc. 104:526-528.
- Carle, F. L., and M. P. Strub. 1978. A new method for estimating population size from removal data. Biometrics 34:621-630.
- Cattaneo, A. 1983. Grazing on epiphytes. Limnol. Oceanogr. 28:124-132.
- Dermott, R. M., J. Kalff, W. Leggett, and J. Spence. 1977. Production of *Chironomus*, *Procladius*, and *Chaoborus* at different levels of phytoplankton biomass in Lake Memphremagog, Quebec-Vermont. J. Fish. Res. Bd. Can. 34:2001-2007.

- Draper, N. R., and H. Smith. 1981. Applied regression analysis. Wiley Publishers, New York. 709 pp.
- Elliott, J. M. 1969. The daily activity patterns of mayfly nymphs (Ephemeroptera). *Journal of Zoology* 155:201-221.
- Filbin, G. I., and J. W. Barko. 1986. Growth and nutrition of submersed aquatic macrophytes. In R. H. Kennedy and R. C. Gunkel, Jr., eds. *Limnological studies at Eau Galle Lake, Wisconsin; Report 2, Special studies and summary. Technical Report E-85-2.* US Army Engineer Waterways Experiment Station, Vicksburg, Miss.
- Kennedy, R. H., ed. 1985. *Limnological studies at Eau Galle Lake, Wisconsin; Report 1, Introduction and water quality monitoring studies. Technical Report E-85-2.* US Army Engineer Waterways Experiment Station, Vicksburg, Miss.
- Lander, D. H. 1982. Effect of naturally senescing aquatic macrophytes on nutrient chemistry and chlorophyll *a* of surrounding waters. *Limnol. Oceanogr.* 27:428-439.
- Mason, D. F., and R. J. Bryant. 1975. Periphyton production and grazing by chironomids in Alderfen Broad, Norfolk. *Freshwater Biol.* 5:271-277.
- McElhone, M. 1982. The distribution of Naiadae (Oligochaeta) in the littoral zone of selected lakes in North Wales and Shropshire. *Freshwater Biol.* 12:421-425.
- Moore, J. W. 1980. Factors influencing the composition, structure, and density of a population of benthic invertebrates. *Archiv fur Hydrobiol.* 88:201-218.
- Ross, H. R. 1944. The caddis flies, or Trichoptera, of Illinois. *Bulletin of the Illinois Natural History Survey* 23:1-326.
- Savino, J. F., and R. A. Stein. 1982. Predator-prey interaction between largemouth bass and blue gills as influenced by simulated, submersed vegetation. *Transactions, American Fishery Society* 111:255-266.
- Stolbunova, V. N., and A. K. Stolbunov. 1980. The natural life of the littoral zone of a reservoir and its effect on the pelagic zone (with reference to the bacterioplankton and zooplankton of Ivan'kovo Reservoir). *Hydrobiological Journal* 16:1-6.
- Street, M., and G. Titmus. 1982. A field experiment on the value of allochthonous straw as food and substratum for lake macroinvertebrates. *Freshwater Biol.* 12:403-410.
- Wetzel, R. G. 1975. *Limnology.* W. B. Saunders Company, Philadelphia, Pa.

Introduction

143. Sedimentation is an important process which results in the accumulation of materials in reservoirs. In addition to losses in water storage capacity, deposited sediment and associated nutrients can also have a direct or indirect impact on reservoir water quality (Thornton et al. 1981). Excessive deposition of material rich in organic material, metals, and nutrients can aggravate hypolimnetic dissolved oxygen demands, thereby promoting the dissolution of constituents which have undesirable characteristics (e.g., sulfur, iron, manganese, organic pollutants). A knowledge of the origins of deposited material, including metals and nutrients, and its potential impact on material cycles will greatly facilitate an improved understanding of reservoir ecosystems.

144. Depositional patterns in reservoirs would be expected to vary, depending on watershed characteristics, basin morphometry, and the hydrology of the system. For instance, in large reservoirs dominated by tributary inflow, deposition rates are strongly influenced by influent nutrient delivery, varying longitudinally with the extent of advective sediment transport (James and Kennedy 1986). Near the inflow region, large deposits of material accumulate in the sediments due to allochthonous loading. Down-reservoir, where advective influences are often negligible, deposition rates often decrease substantially and reflect more localized conditions within the reservoir. The displacement of allochthonous nutrients to the headwater sediments can, therefore, have a marked effect on nutrient loads to down-reservoir locations (Kennedy et al. 1983).

145. In lakes and reservoirs less influenced by river loading, deposition is more likely influenced by processes occurring within the system. In productive lakes, deposition of algae often corresponds

* Part VII was written by William F. James.

AD-A182 298

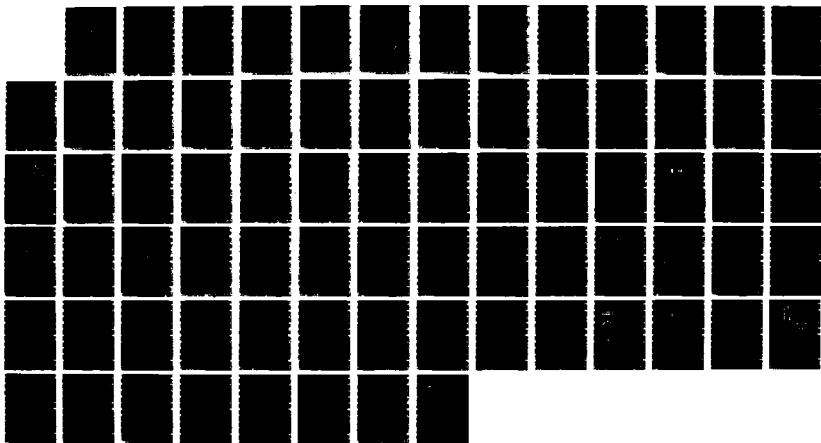
ENVIRONMENTAL AND WATER QUALITY OPERATIONAL STUDIES
LIMNOLOGICAL STUDIES (U) ARMY ENGINEER WATERWAYS
EXPERIMENT STATION VICKSBURG MS ENVIR

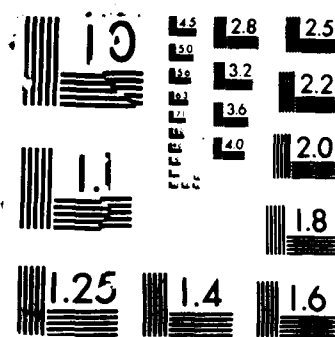
2/2

UNCLASSIFIED

R H KENNEDY ET AL MAR 87 WES/TR/E-85-2-2 F/G 8/8

NL





seasonal algal periodicity, indicating that autochthonous sources are an important component in sedimentary flux (Fallon and Brock 1980; Livingstone and Reynolds 1981; Reynolds, Morison, and Butterwick 1982). The deposition of decomposing algae is also an important factor in nutrient cycling in the water column (Bloesch, Stadelmann, and Buhrer 1977; Kimmel and Goldman 1977) and subsidizes benthic detrital dynamics, thereby providing a link between productivity and decomposition (Hargrave 1973).

146. Seasonal mixing in shallow lakes would also have an effect on depositional patterns and the distribution of nutrients associated with the sediment. In lakes and reservoirs experiencing periods of turnover, sediment may be resuspended and redistributed from the littoral areas where it is then deposited in the deeper portions of the lake. The result is the gradual accumulation of sediment and nutrients to the deepest sediments. Davies (1973) observed this depositional feature (sediment focusing) in a study of pollen grain distribution in Frains Lake, Michigan. Water circulation also causes the reintroduction of dissolved oxygen to the hypolimnion, causing additional deposition of precipitated metals (Davison, Woof, and Rigg 1982) and phosphorus (Kennedy et al. 1983) to the deeper portion of the lake.

147. The detection of seasonal and spatial fluctuations in deposition can, therefore, provide valuable information concerning lake and reservoir dynamics. This study was conducted to examine depositional patterns in Eau Galle Lake in an effort to better define the origins and fate of settling material.

Methods

148. Sampling stations chosen to characterize water quality patterns in the reservoir are discussed in Johnson and Carroll (1985). Four of these sampling locations were chosen to characterize temporal and spatial patterns in sedimentation rates. These were: station 40 (2-m depth), located near the river mouth region; stations 30 and 60 (less than 4-m depth), positioned in shallow areas along the eastern and

western shorelines; and station 20 (9-m depth), located at the deepest area of the reservoir.

149. Sedimentation traps used were those described in James and Kennedy (1986). The traps were designed to efficiently collect settling material and had a cylindrical shape (Hargrave and Burns 1979) and an aspect ratio of 3.0 (Lau 1979). Three sedimentation traps, suspended on a carousel system, were deployed 0.5 m above the sediment surface at sampling locations mentioned above. An additional carousel was suspended at the 4-m depth at station 20 (metalimnetic area). Before deployment, traps filled with lake water were treated with 1 l of a 5-percent salt solution and mercuric chloride (750 µg/l) to reduce potential resuspension and decomposition, respectively. Settled material was removed at 2-week intervals during the summer months (April through October) and monthly during ice cover in winter.

150. Sample analyses were begun immediately following collection. Collected material was strained through 300-µ-mesh net to remove large zooplankton when present. Noticeable quantities of zooplankton were observed in the settled material only during vernal and autumnal circulation. Remains were dead, apparently killed by the salt solution, suggesting that zooplankton did not graze on the collected material. Removal of zooplankton did not appear to measurably affect sedimentation rates (i.e., accounted for less than 2 percent of the total dry weight collected).

151. Sample aliquots were obtained during thorough mixing and analyzed for seston, organic carbon, nitrogen, phosphorus, iron, manganese, and algal pigments. Chemical analyses of the settling material followed standard methods (American Public Health Association 1980) and methods discussed by Johnson and Lauer (1985). Concentrations were converted to sedimentation rates by the equation

$$R = (C \cdot V) / (D \cdot A)$$

where

- R = sedimentation rate ($\text{mg}/\text{m}^2/\text{day}$)
- C = material concentration (mg/ℓ)
- V = volume of collected sediment (ℓ)
- D = deployment (days)
- A = trap mouth area (m^2)

Results

152. Deposition in Eau Galle Lake was strongly influenced by flow patterns, algal blooms, and mixing events. Trap data are presented in detail from stations 20 and 40. Where appropriate, data from stations 30 and 60 will be discussed.

153. Eau Galle Lake received the majority of its water income from the Eau Galle River (Kennedy 1986). During the 2-year study period, major discharge peaks in mid-February 1981 and early-April 1982 corresponded to spring snowmelt and runoff. The greatest discharge event occurred during the snowmelt of 1982, reaching a maximum of $2,091 \text{ m}^3/\text{sec}$. Discharge peaks in April and May of both years were associated with spring storm events. Flows were lowest from June through September with means of less than $10 \text{ m}^3/\text{sec}$ in 1981 and 1982. However, some notable, smaller storm-related inflows occurred in mid-June, August, and mid-September 1981. In general, flows associated with snowmelt periods accounted for a majority of the annual discharges of both years (Kennedy 1986). Summer discharges contributed little to the annual water budget in 1981 and 1982.

154. Sedimentation rates, as measured in traps, displayed marked spatial patterns during the snowmelt period of early April 1982 (Table 10). Traps situated in the deepest area of the reservoir (i.e., station 20) exhibited the highest seston, organic carbon, phosphorus, and iron deposition rates during this 4-day period of elevated flow. Deposition along the shallow littoral areas at stations 30 and 60 was minimal. Sedimentation rates were also low in the river mouth region at station 40, where turbulent flow was probably greatest. These patterns

Table 10
Spatial Variations in Deposition During a Snowmelt
and Runoff Period, April 1982

Variable	Station (Depth)			
	20 (8m)	30 (4m)	40 (2m)	60 (2m)
Seston ($\text{g/m}^2 \cdot \text{day}$)	913	31	109	15
Organic carbon ($\text{mg/m}^2 \cdot \text{day}$)	52,000	2,000	5,000	1,000
Phosphorus ($\text{mg/m}^2 \cdot \text{day}$)	900	60	140	20
Iron ($\text{mg/m}^2 \cdot \text{day}$)	18,700	790	2,120	260

indicate that material loads were deposited primarily in the deepest areas of the reservoir.

155. A comparison of seasonal patterns in chlorophyll a concentrations (Figure 25) and deposition (Figure 26) indicated that autochthonous material was an important source of deposited material since chlorophyll a deposition closely corresponded to the occurrence of algal blooms. At the 4-m depth at station 20, a pronounced depositional peak occurred in early-May 1981, 2 weeks after a chlorophyll a maximum was observed in April. Other depositional increases at the 4-m depth occurred in mid-July, late-September, and late-October 1981, again following major epilimnetic chlorophyll a increases. In 1982, similar trap rate increases followed major blooms. At the 8-m depth, deposition maxima in chlorophyll a were not always evident following chlorophyll a increases in the water column. While pronounced deposition increases occurred in May of both years, coincident with increases observed at the 4-m depth, trap rates at the 8-m depth did not appear to respond to maxima in chlorophyll a concentrations in early-July of both years. Notable increases occurred in early-October of both years but lagged behind deposition maxima at the 4-m depth by 2 weeks.

156. At station 40, seasonal patterns of chlorophyll a deposition were also in close agreement with seasonal maxima in chlorophyll a in the water column (Figures 25 and 27). Deposition peaks in early-May,

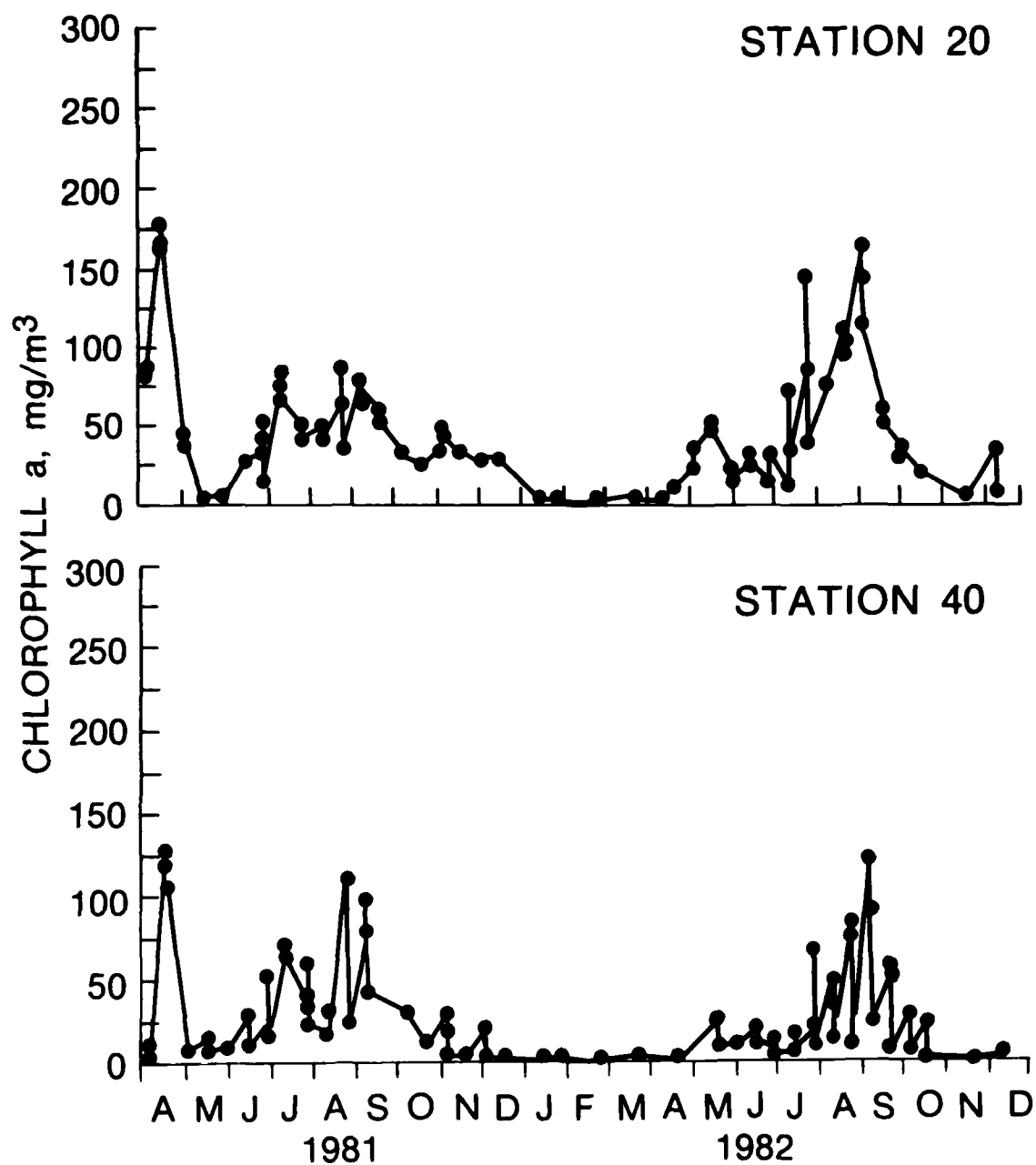
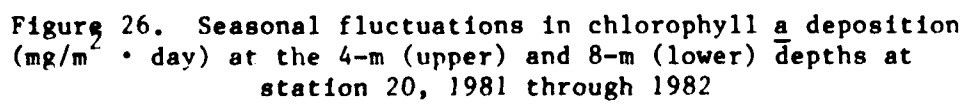


Figure 25. Changes in average chlorophyll a (mg/m³) concentrations in the euphotic zone (0 to 2 m) at stations 20 (upper) and 40 (lower)



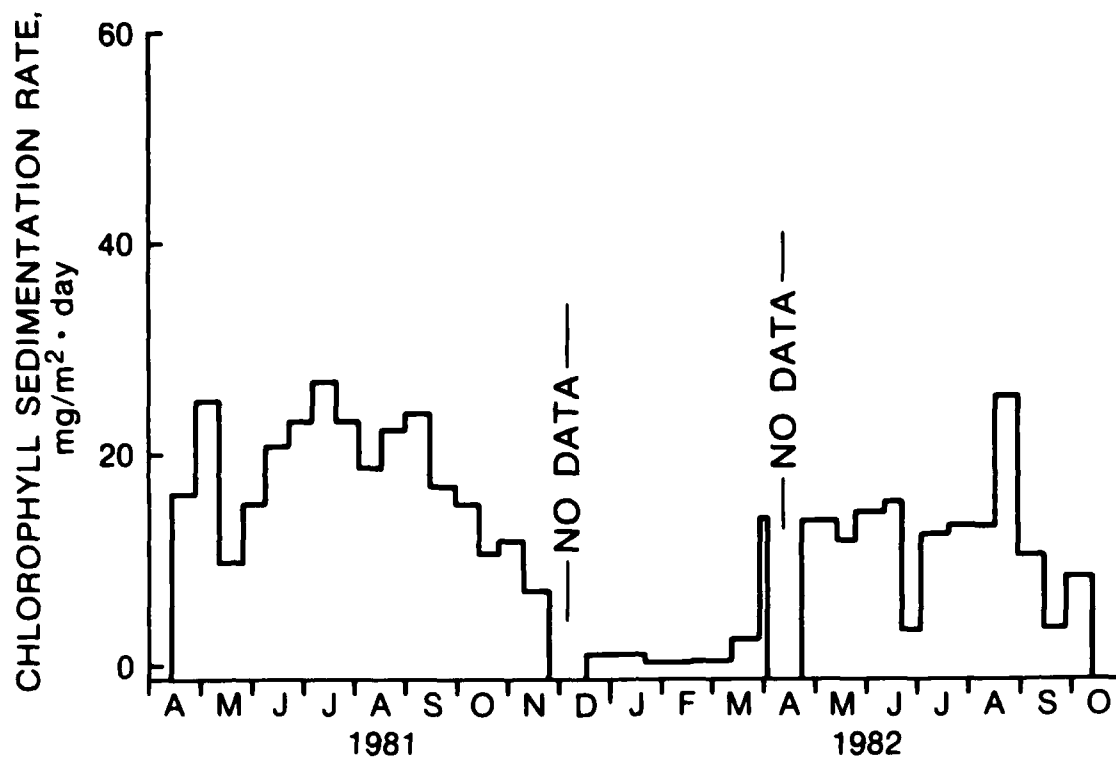


Figure 27. Seasonal fluctuations in chlorophyll a deposition ($\text{mg/m}^2 \cdot \text{day}$) at station 40, 1981 through 1982

July, and early-September corresponded to increased water-column concentrations in April, early-July, and August, respectively. Similar relationships between concentration increases and deposition were also evident in 1982.

157. Spatial and seasonal patterns of organic carbon deposition generally coincided with chlorophyll a, suggesting that algal remains were an important component of carbon flux to the sediment. At station 20, organic carbon deposition maxima were observed at the 4-m depth from April through early-May, and again in late-July (Figure 28). These corresponded to pronounced increases in chlorophyll a concentrations in the water column and chlorophyll a depositional peaks. Smaller trap rate increases were noted in late-September and late-October, then rates were substantially lower during the winter months. In 1982, major peaks in carbon deposition in May and September reflected trap rate patterns of chlorophyll a.

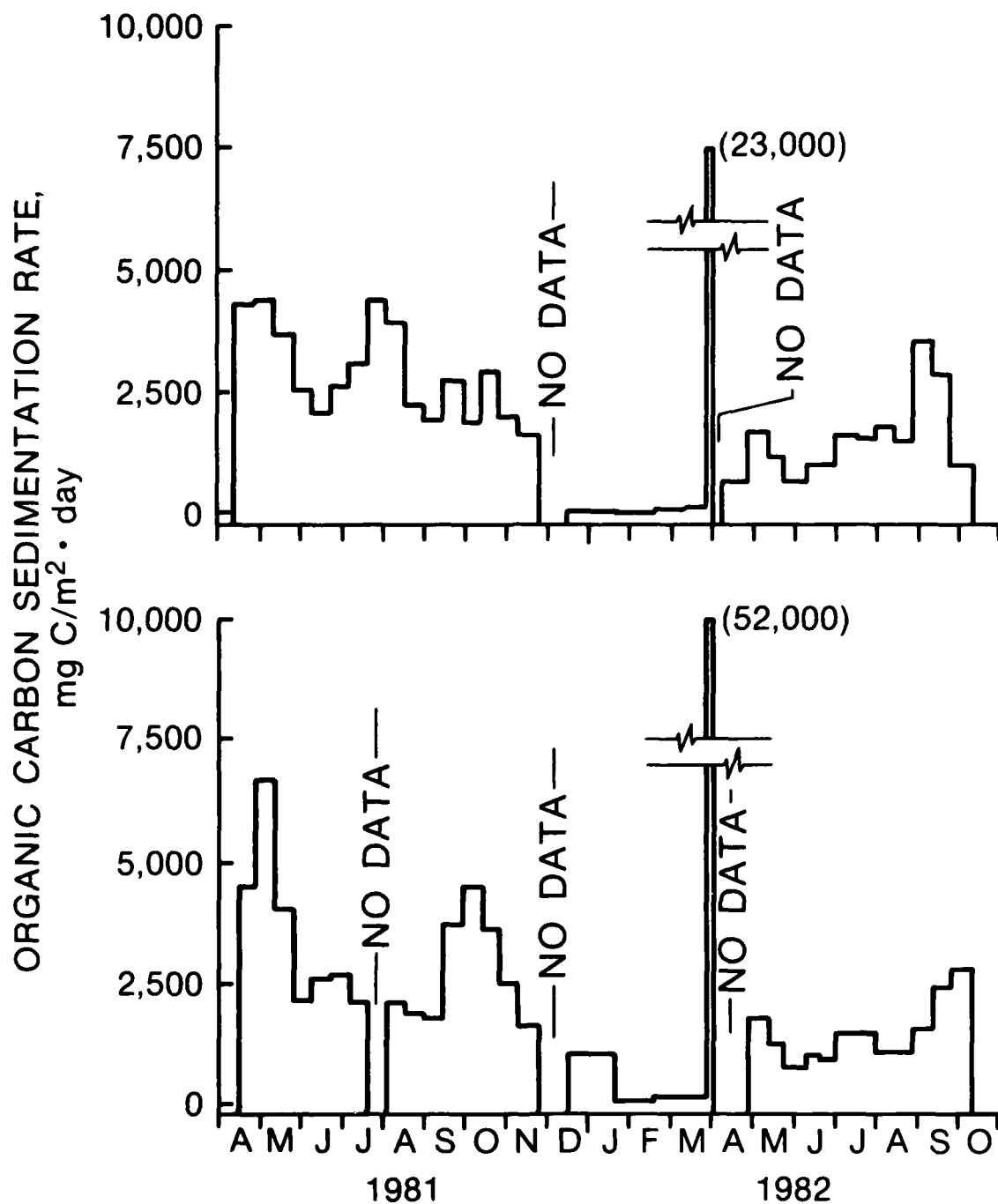


Figure 28. Seasonal fluctuations in organic carbon deposition (mg/m² · day) at the 4-m (upper) and 8-m (lower) depths at station 20

158. At the 8-m depth, depositional peaks in May of both years corresponded with rate increases observed at the 4-m depth. However, major peaks in October of both years appeared to lag behind depositional peaks at the 4-m depth by 2 weeks. Organic carbon deposition did not exhibit distinct fluctuations at the 8-m depth in July 1981 and September 1982, although distinct depositional peaks in organic carbon and chlorophyll a were evident at the 4-m depth during this period.

159. At station 40, organic carbon deposition exhibited summer maxima during both years (Figure 29). Distinct peaks were not apparent from June through August 1981; however, depositional peaks in July and August of 1982 corresponded to peaks in chlorophyll a deposition.

160. During the 2-year period, seston deposition exhibited marked temporal and spatial variations which appeared to be related to external loading as well as events occurring within the lake. In April and early-May 1981, trap rates at station 20 increased substantially at the 8-m depth coincident with a distinct peak in chlorophyll a concentration and an early-May freshet (Figure 30). A second peak in seston deposition occurred at the 8-m depth in mid-June 1981, which did not correspond to the occurrence of an algal bloom, but rather to a small, mid-June freshet. As will be discussed later, river loading during this period may have influenced deposition rates at the 8-m depth. Seston deposition rates at the 4-m depth, which were lower than those at 8 m, did not fluctuate in June but exhibited increases in October of both years which coincided with fall turnover. At the 8-m depth, pronounced depositional increases were also apparent during fall turnover.

161. Near the shallow river mouth area (station 40), peaks in seston deposition were more pronounced and of a greater magnitude (Figure 31). Major peaks occurred in early-May and June 1981, coinciding with those at station 20. Rates were low from mid-July through early-August 1981; then, a deposition peak occurred in mid-August, which did not correspond to depositional variations at station 20. After minimal rates during the winter, major trap rate increases were observed in May, late-June, and late-August 1982.

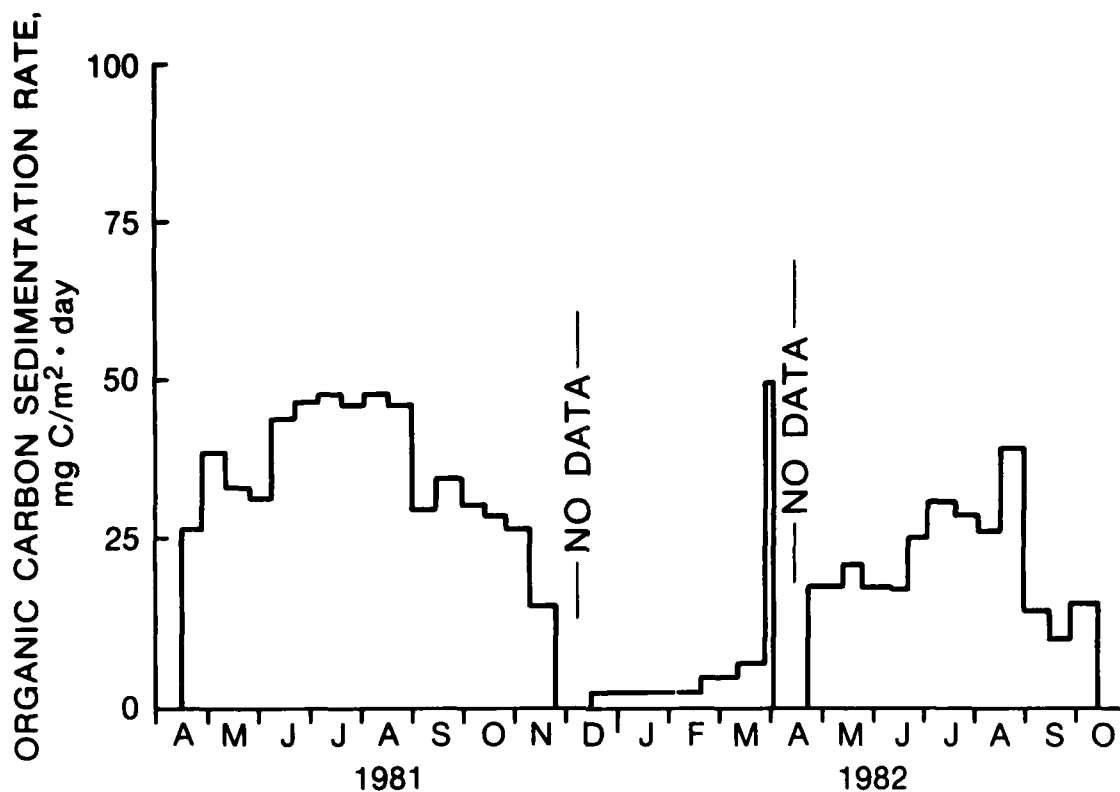


Figure 29. Seasonal fluctuations in organic carbon deposition ($\text{mg}/\text{m}^2 \cdot \text{day}$) at station 40

162. Iron trap rates exhibited seasonal and spatial trends which were similar to those for seston. Major depositional increases occurred at the 4-m depth at station 20 in May, October, and November 1981 (Figure 32). Deposition declined during the winter period, then increased to a rate of $10 \text{ g}/\text{m}^2 \cdot \text{day}$ in early-April 1982. During summer 1982, depositional increases occurred from April through May, and September through October. At the 8-m depth at station 20, depositional variations were generally similar to the seasonal patterns exhibited at the 4-m depth, but of a greater magnitude. A notable peak in iron deposition was evident in late-June 1981 at the 8-m depth which was not reflected in depositional patterns at 4 m. Major peaks in deposition in October of both years coincided with fall turnover. At station 40, iron deposition exhibited marked increases in June, August, and October 1981 (Figure 33). Peaks in iron deposition were less distinct in 1982; however, small increases occurred in early-May, June, August, and October.

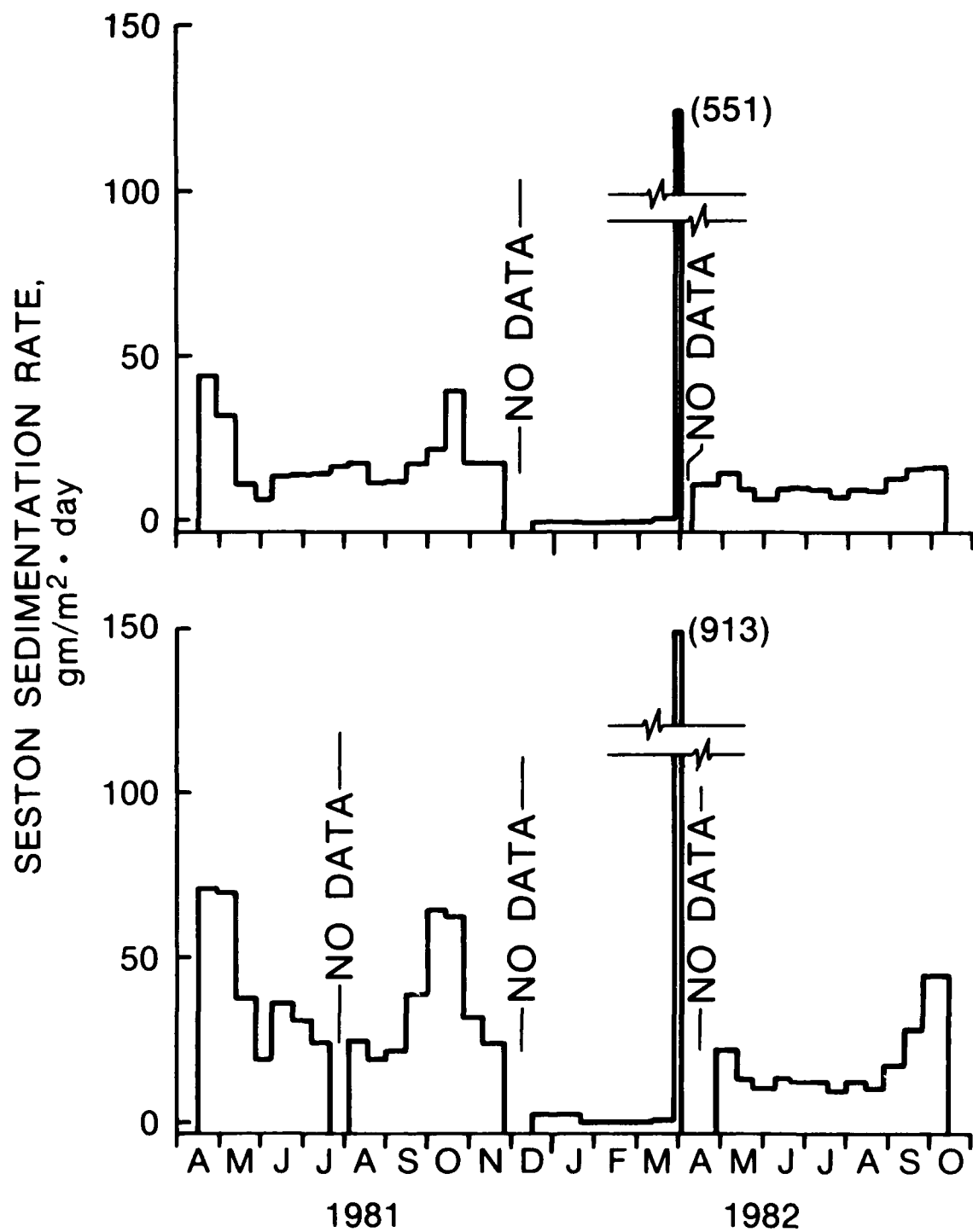


Figure 30. Seasonal fluctuations in seston deposition (g/m² · day) at station 20 at the 4-m (upper) and 8-m (lower) depths

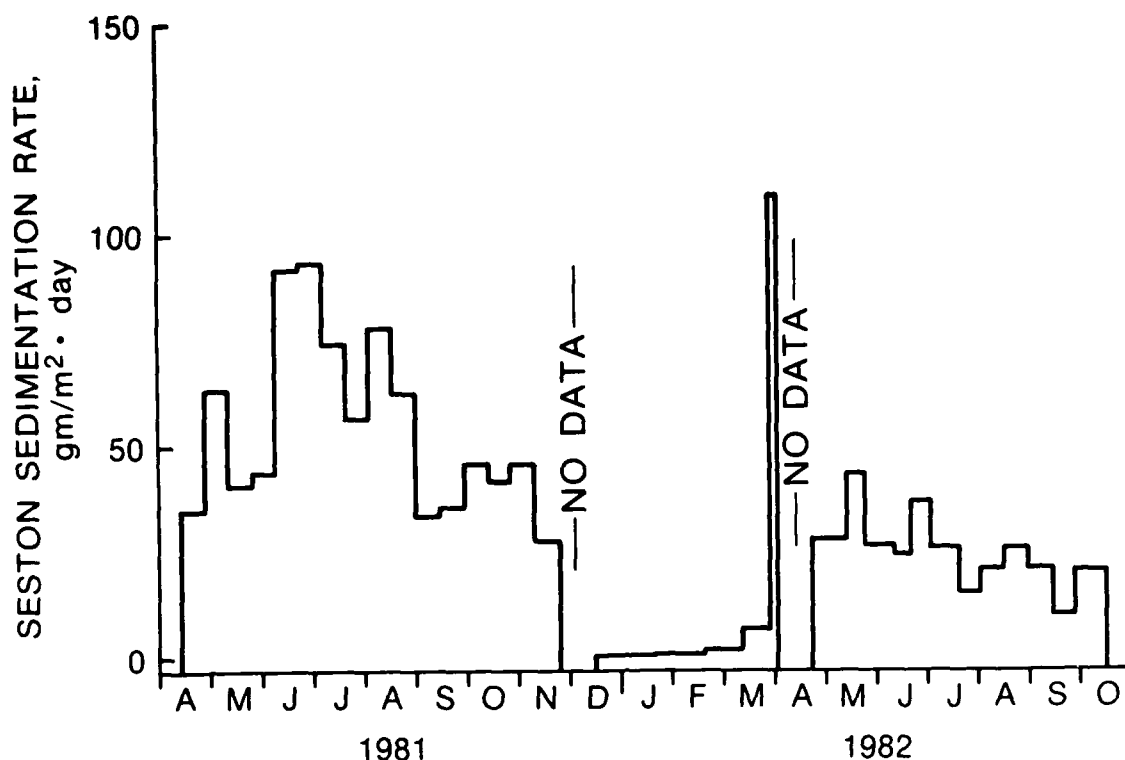


Figure 31. Seasonal fluctuations in seston deposition ($\text{g/m}^2 \cdot \text{day}$) at station 40

163. Phosphorus deposition exhibited a complex spatial and seasonal pattern. In general, major depositional increases coincided with algal blooms, high flows, and fall turnover. In 1981, phosphorus deposition increased substantially at station 20 at the 4- and 8-m depths in April and May (Figure 34), during a period of peak chlorophyll a concentration in the water column. The fact that chlorophyll a deposition increased during this period, as well, indicated that major phosphorus deposition was associated with algal remains. A second deposition occurred in June 1981 at both depths, when algal biomass and chlorophyll a deposition were low. It was believed that these peaks were the result of phosphorus loading from a small, mid-June freshet. It was noted that deposition was greatest at the 8-m depth in June, suggesting that phosphorus sedimentation occurred primarily in the hypolimnion. A depositional increase occurred at the 4-m depth at station 20 in late-July,

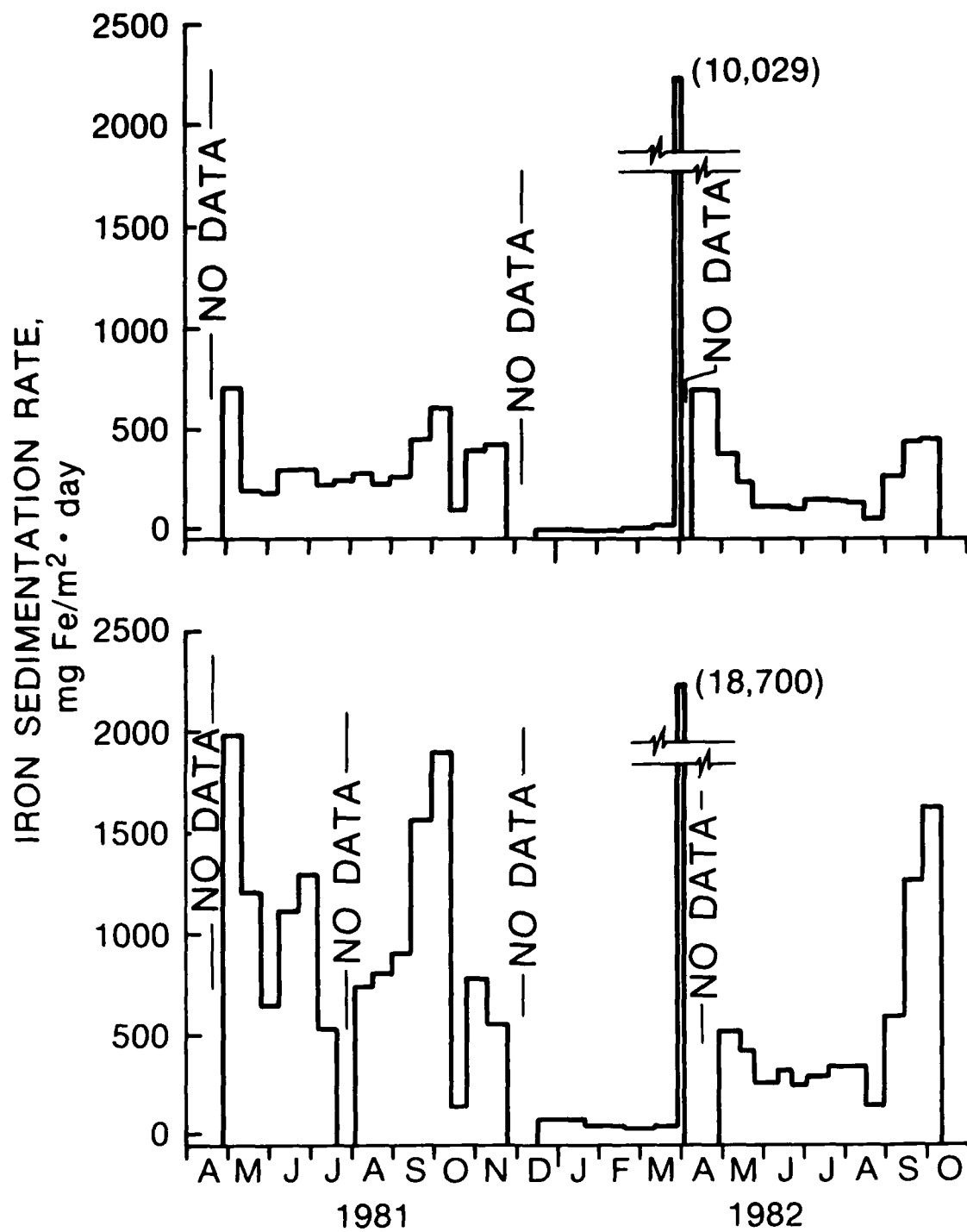


Figure 32. Seasonal fluctuations in iron deposition (mg/m² · day) at station 20 at the 4-m (upper) and 8-m (lower) depths

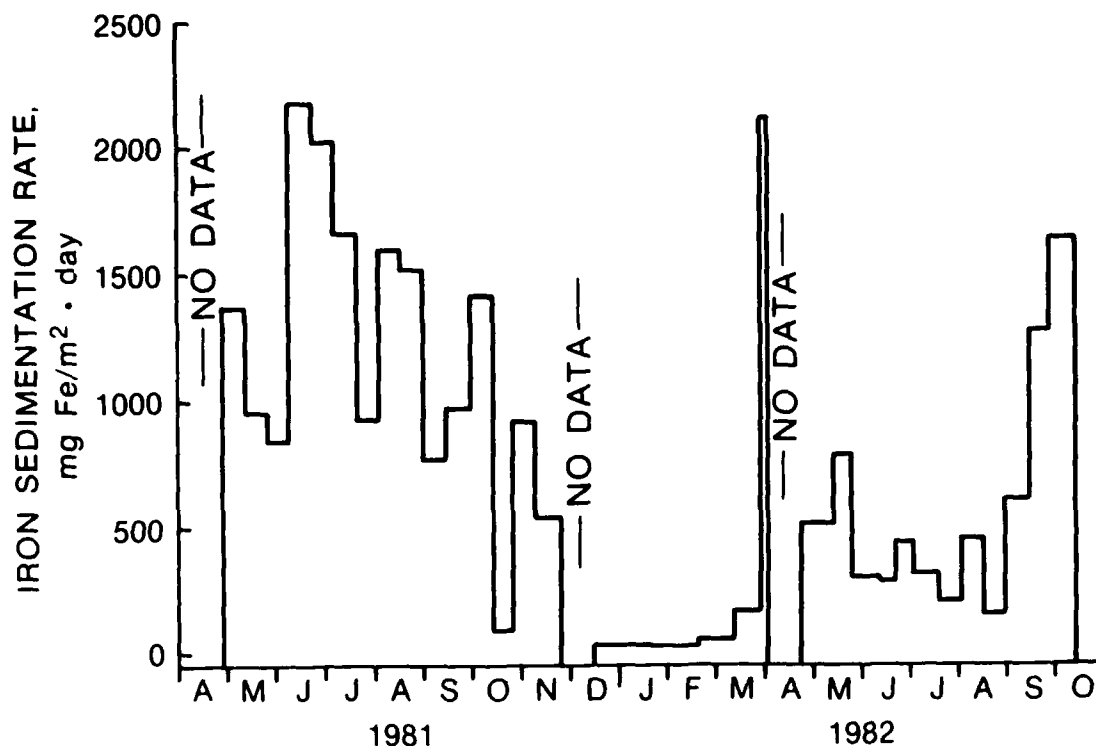


Figure 33. Seasonal fluctuations in iron deposition ($\text{mg}/\text{m}^2 \cdot \text{day}$) at station 40

which corresponded with an algal bloom in the water column. Deposition at the 8-m depth did not increase until September 1981. Since this increase at 8 m occurred during thermocline erosion and turnover, it suggested the possible chemical precipitation of phosphorus from the hypolimnion to the sediment.

164. In 1982, phosphorus deposition rates at station 20 were generally lower in magnitude than 1981 values. At the 4-m depth, depositional peaks in May, July through early-August, and September corresponded with peaks in chlorophyll a concentrations in the water column and chlorophyll a deposition. Trap rate increases at the 8-m depth occurred in May and late-September.

165. At station 40, deposition maxima were evident primarily during the summer months of both years (Figure 35). Rate increases in early-May and July 1981 corresponded with increases in chlorophyll a deposition, suggesting that algae were a possible origin to the settling

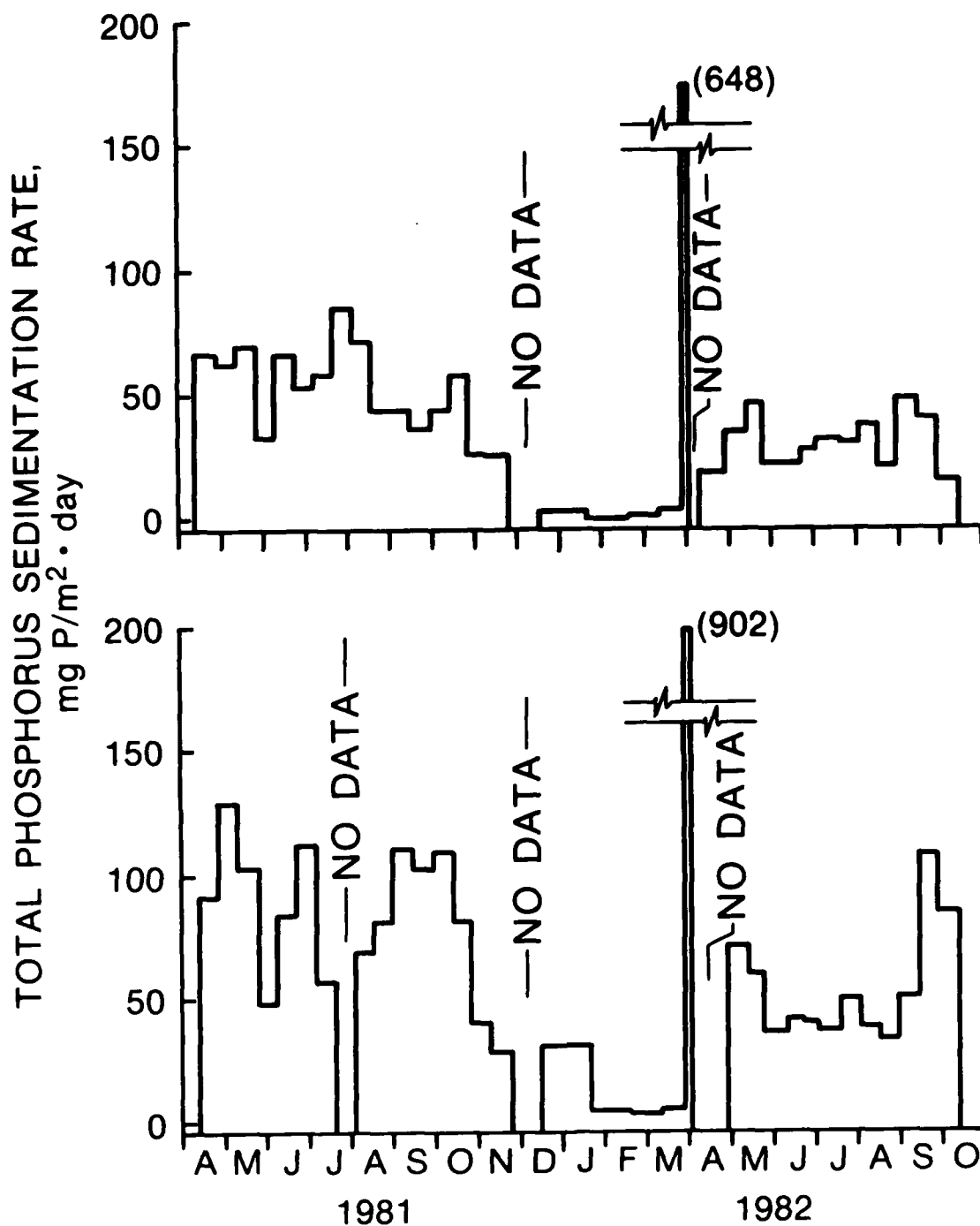


Figure 34. Seasonal fluctuations in phosphorus deposition (mg/m² · day) at station 20 at the 4-m (upper) and 8-m (lower) depths

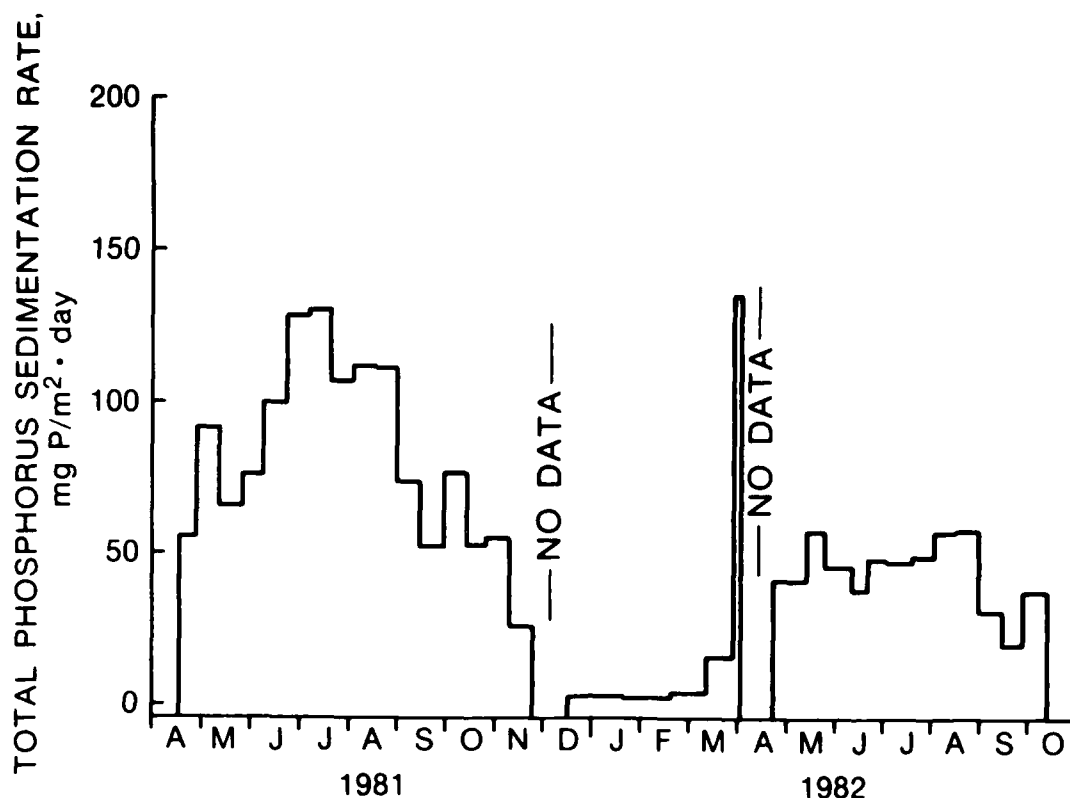


Figure 35. Seasonal fluctuations in phosphorus deposition ($\text{mg}/\text{m}^2 \cdot \text{day}$) at station 40

material during this period. Deposition remained high through August 1981. During the summer of 1982, peaks in deposition were less distinct; however, elevated rates from late-June through August coincided with peaks in chlorophyll *a* concentrations in the water column.

Discussion

166. Spatial and temporal patterns in deposition were related to: (a) snowmelt and runoff, (b) algal blooms, and (c) water column mixing events. Deposition increases during these events were marked and provided a general pattern for sedimentation dynamics and the distribution of sediment in the reservoir.

167. In winter, river loading during periods of snowmelt had a pronounced effect on material deposition. For instance, deposition during a snowmelt period in 1982 accounted for 63 percent of the annual

seston deposition at station 20 at the 8-m depth in 1982. Similar comparisons could be made with organic carbon, phosphorus, and iron. Trap data indicated that allochthonous loads were primarily deposited in the deepest portion of the reservoir. Lower deposition rates at station 40 suggested that riverborne particles entering the reservoir remained suspended in the river mouth region, settling at midreservoir as advective velocities decreased. Trap rates were lower at stations 30 and 60, and spatial patterns suggested that river water moved directly to the out-flow structure, bypassing littoral areas. Since trap rates were maximal during the snowmelt period, preferential deposition of iron and phosphorus to the middle of the reservoir would supply nutrients to the sediment which would be potentially resolubilized during summer hypolimnetic anoxia. Annually, high deposition rates during snowmelt periods would act to focus elements important in nutrient cycles to midreservoir.

168. Apparent during the summer months of both years (April through October) were marked similarities between fluctuations in chlorophyll a concentrations in the water column and the deposition of chlorophyll a. The data indicated that major inputs of algal remains could be clearly distinguished on a seasonal basis. Other recent studies have also found close relationships between algal periodicity and deposition, inferring that sedimentation is a major process of loss from the water column to some species. For instance, Reynolds, Morison, and Butterwick (1982) found good agreement between diatom standing crop and deposition, suggesting that sedimentation is a significant means of loss of this algal group. In Eau Galle Lake, Barko et al. (1986) reported that diatoms were the major planktonic species during May of both years. Chlorophyll a trap data suggest substantial sedimentation of diatoms after peaks in their biomass during the spring of each year. Nutrients associated with these blooms were organic carbon, phosphorus, and iron, suggesting that diatoms are major nutrient sources to the sediment. Barko et al. (1986) associated peaks in chlorophyll a concentrations during the summer months of 1981 and 1982 (July through September) with the occurrence of *Ceratium* sp. At station 20, chlorophyll a deposition

increased at the 4-m depth during chlorophyll a concentration increases in July and September 1981 and 1982, suggesting that *Ceratium* remains may have comprised a significant portion of the settling material. However, chlorophyll a deposition maxima were not always apparent at the 8-m depth during these periods. Furthermore, in July of both years, chlorophyll a deposition decreased with depth from the 4-m to the 8-m depth. Reasons for these depth-related differences included possible vertical migration of *Ceratium* into the 4-m trap but not into the 8-m trap and/or decomposition of algal remains in the water column.

169. Small summer freshets during 1981 also appeared to influence depositional patterns. For instance, in June 1981, seston, phosphorus, and iron deposition increased substantially at stations 20 and 40. The peaks could not be identified with settling algae, because chlorophyll a concentrations in the water column and chlorophyll a deposition were low during this period. The peaks, however, did coincide with a small freshet (16 June 1981) which caused large increases in seston, iron, and phosphorus concentrations in the river mouth region (Johnson and Carroll 1985). Similar increases in iron and seston deposition occurred at station 40 during an August 1981 freshet. However, deposition rates did not fluctuate at station 20 during this period.

170. These results suggested that particulate material loads during small summer freshets were retained, to a large extent, in the reservoir. Although peaks in the hydrograph, and loading, were small during the summer (Kennedy 1986), the flushing rate was lower, resulting in particle deposition primarily near the river mouth and, to a lesser extent, at midreservoir. These spatial patterns contrasted markedly with patterns of deposition during spring snowmelt, when loading and the flushing rate were high (Ashby and James 1985). Thus, it appears that maxima in deposition shift spatially from midreservoir to the river mouth in response to the magnitude of loading and the hydraulic residence time.

171. It also appeared that small summer freshets had an impact on annual variations in iron and seston deposition. For instance, deposition at station 40 was lower and did not exhibit marked fluctuations in

summer 1982, compared to the more pronounced fluctuations in summer 1981. Coincident with these seasonal patterns was a lack of occurrence of summer freshets in 1982. The thermal nature of Eau Galle Lake provided a mechanism for potential sediment resuspension, chemical precipitation, and deposition in the deeper areas during autumnal turnover. During periods of turnover, the deeper areas of the reservoir acted as zones of accumulation, particularly for iron and phosphorus. Thermocline erosion and mixing in the fall resulted in the introduction of dissolved oxygen to the hypolimnion (Johnson and Carroll 1985), thereby causing the coprecipitation of iron and phosphorus. Precipitation processes probably did not occur in the well-oxygenated shallow areas; hence, depositional increases of iron and phosphorus in these regions were probably the result of resuspension during turnover. Gunkel et al. (1984) found metal and nutrient concentrations were highest in the deepest sediment of Eau Galle Lake, further suggesting that this region acted as a nutrient sink.

172. Seston deposition also exhibited marked spatial patterns during turnover, which were related to water-column mixing. Trap rates were greatest at station 20 (suggesting possible additional sediment resuspension) but were less pronounced at the shallow station. Considering the shallow basin morphology and the potential erosional aspects of the littoral areas (Gunkel et al. 1984), these spatial patterns suggested sediment redistribution within the reservoir during turnover periods (i.e., sediment focusing). For instance, Davies (1973) found that pollen grains in the littoral sediments of Frains Lake, Michigan, were resuspended and redistributed during turnover periods, resulting in greater accumulation rates in the deeper portions of the lake, where turbulent resuspension was less evident. The findings of Gunkel et al. (1984) suggested that sediment focusing was a potentially important mechanism in the spatial distribution of sediment particle sizes in Eau Galle Lake.

Conclusions

173. In general, seasonal and spatial differences in trapping rates appeared to be strongly influenced by basin morphology, hydrologic events, and algal biomass. The small size and shallow nature of Eau Galle Lake result in the accumulation of sediment in the deepest area of the reservoir. During spring snowmelt, nutrient incomes are trapped primarily in the deepest portion of the reservoir, promoting the preferential accumulation of ecologically important nutrients in this region. During the summer stratified period, small summer freshets appear to have a localized influence on depositional patterns, and nutrient loads are primarily deposited near the inflow area. Deposition is also mediated by biological events in summer. In fall, water-column mixing associated with turnover causes chemical precipitation and accumulation of nutrients to the deep sediments as well as potential redeposition of sediments from shallow to deeper portions of the reservoir.

174. Inferences from this study might provide insight to sedimentation processes occurring in other small lakes and reservoirs with shallow basin morphologies. While external loading appears to be an important source of sediment to reservoirs, mixing processes may also have a pronounced effect on depositional patterns and the distribution of sediment as basin morphology decreases. In lakes and reservoirs experiencing sediment resuspension and redistribution, the focusing of metals and nutrients to deeper sediment, and their release during anoxia, may have a bearing on biological and chemical events occurring in the water column.

References

American Public Health Association. 1980. Standard methods for the examination of water and wastewater, 15th ed. Washington, DC.

Ashby, S. A., and W. F. James. 1985. Limnology of Eau Galle tributaries. In R. H. Kennedy, ed. Limnological studies at Eau Galle Lake, Wisconsin; Report 1, Introduction and water quality monitoring studies. Technical Report E-85-2. US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Barko, J. W., D. J. Bates, G. J. Filbin, S. M. Hennington, and D. G. McFarland. 1986. Seasonal growth and community composition of phytoplankton. In R. H. Kennedy and R. C. Gunkel, Jr., eds. Limnological studies at Eau Galle Lake, Wisconsin; Report 2, Special studies and summary. Technical Report E-85-2. US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Bloesh, J., P. Stadelmann, and H. Buhner. 1977. Primary production, mineralization, and sedimentation in the euphotic zone of two Swiss lakes. *Limnol. Oceanogr.* 22:511-526.

Davies, M. D. 1973. Redeposition of pollen grains in lake sediments. *Limnol. Oceanogr.* 18:44-52.

Davison, W., C. Woof, and E. Rigg. 1982. The dynamics of iron and manganese in a seasonally anoxic lake; direct measurement of fluxes using sediment traps. *Limnol. Oceanogr.* 27:987-1003.

Fallon, R. D., and T. D. Brock. 1980. Planktonic blue-green algae: production, sedimentation, and decomposition in Lake Mendota, Wisconsin. *Limnol. Oceanogr.* 25:72-88.

Gunkel, R. C., Jr., R. F. Gaugush, R. H. Kennedy, G. E. Saul, J. H. Carroll, and J. E. Gauthey. 1984. A comparative study of sediment quality in four reservoirs. Technical Report E-84-2. US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Hargrave, B. T. 1973. Coupling carbon flow through some pelagic and benthic communities. *J. Fish. Res. Board Can.* 30:1317-1326.

Hargrave, B. T., and N. M. Burns. 1979. Assessment of sediment trap collection efficiency. *Limnol. Oceanogr.* 24:1124-1136.

James, W. F., and R. H. Kennedy. 1986. Patterns of sedimentation at DeGray Lake, Arkansas. In R. H. Kennedy and J. Nix, eds. Proceedings of the DeGray Lake Symposium. Technical Report in preparation. US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Johnson, D. R., and J. H. Carroll. 1985. Physicochemical limnology of Eau Galle Lake. In R. H. Kennedy, ed. Limnological studies at Eau Galle Lake, Wisconsin; Report 1, Introduction and water quality monitoring studies. Technical Report E-85-2. US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Johnson, D. R., and G. J. Lauer. 1985. General methods. In R. H. Kennedy, ed. Limnological studies at Eau Galle Lake, Wisconsin; Report 1, Introduction and water quality monitoring studies, Technical Report E-85-2. US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Kennedy, R. H. 1986. Material loadings to Eau Galle Lake. In R. H. Kennedy and R. C. Gunkel, Jr., eds. Limnological studies at Eau Galle Lake, Wisconsin; Report 2, Special studies and summary. Technical Report E-85-2. US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Kennedy, R. H., R. H. Montgomery, W. F. James, and J. Nix. 1983. Phosphorus dynamics in an Arkansas reservoir: the importance of seasonal loading and internal recycling. Miscellaneous Paper E-83-1. US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Kimmel, B. L., and C. R. Goldman. 1977. Production, sedimentation, and accumulation of particulate carbon and nitrogen in a sheltered subalpine lake. Pages 148-155 in Interactions between sediments and freshwater. Pudoc, Amsterdam.

Lau, Y. L. 1979. Laboratory study of cylindrical sedimentation traps. J. Res. Board Can. 36:1288-1291.

Livingstone, D., and C. S. Reynolds. 1981. Algal sedimentation in relation to phytoplankton periodicity in Rostherne Mere. Br. Phycol. J. 16:195-206.

Reynolds, C. S., H. R. Morison, and C. Butterwick. 1982. The sedimentary flux of phytoplankton in the south basin of Windermere. Limnol. Oceanogr. 27:1162-1175.

Thornton, K. W., R. H. Kennedy, J. H. Carroll, W. W. Walker, and S. A. Ashby. 1981. Reservoir sedimentation and water quality - an heuristic model. In H. G. Stefan, ed. Proceedings of a Symposium on Surface Water Impoundments, Amer. Soc. Civil Eng., New York.

PART VIII: SEDIMENT DISTRIBUTION AND QUALITY*

Introduction

175. The transport and deposition of sediment to a reservoir is regulated by a number of factors. These include basin morphology, hydrology, and the influent material settling characteristics. Sedimentary conditions in "typical" mainstream reservoirs are most likely dominated by advective transport. These reservoirs are often long and relatively narrow, with an upper riverine zone of high-flow velocities and turbulence. Velocities and turbulence decrease as the reservoir widens and deepens, resulting in longitudinal gradients of sediment accumulation and particle size. Gunkel et al. (1984) observed that expected longitudinal gradients are confounded by preimpoundment conditions and secondary tributaries. Reservoirs less influenced by flow may be more lakelike in morphology and exhibit sediment deposition by focusing. Sediment focusing is defined as the accumulation of fine particulate matter in the deepest basins of a lake. Davis (1973) and Davis and Brubaker (1973) found that initially smaller pollen grains were preferentially deposited on littoral sediments, but during fall circulation resuspension and deposition led to a higher net accumulation rate in the deepest sediments. Net movement of small particles from the littoral to the deeper portions of Lawrence Lake have also been reported by Wetzel et al. (1972).

176. Hakanson (1977) hypothesized that: (a) fine particulate matter will not be deposited in "high energy environments" (i.e., littoral and turbulent areas), (b) deposition of all particulate matter will be primarily influenced by hydrological flow patterns and bottom topology, and (c) the rate of deposition will increase with increasing depth. His studies also demonstrated a correspondence between sediment distribution and sediment moisture content. Sediments in the transport

* Part VIII was written by Robert C. Gunkel, Jr., Robert F. Gaugush, and Robert H. Kennedy.

and erosion zones of Lake Vanern, Sweden, were found to have moisture contents of 40 to 50 percent, while those in the accumulation zones had moisture contents of 60 to 75 percent. In addition, Hakanson (1977) observed that concentrations of nutrients and metals associated with particulate matter varied proportionally with moisture content.

177. A survey of sediment quality at Eau Galle Lake was conducted to provide information concerning potential relationships between sediment characteristics and reservoir morphometry, hydrology, and water quality. Presented here are observed depositional patterns and a discussion of patterns in sediment quality.

Methods and Materials

178. Sediment core samples were collected at Eau Galle during the period 1-2 February 1980. Sample stations were located so as to coincide with those established during previous water quality studies or to incorporate site-specific characteristics. A total of 35 stations were selected for sampling; however, gravel prevented sample collection at 10 stations (Figure 36). Eighteen of the remaining 25 stations provided sufficient material for both particle size and chemical analysis, while six stations are represented by only particle size data. A single station is represented by only chemical data.

179. Sediment samples were collected using a single-barrel Wildco Core Sampler (Wildco Supply Co., Saginaw, Mich.) fitted with polyethylene liners. The core sampler provided a means for identifying surficial sediments and maintaining sample integrity. The sampling involved collecting two core samples from each station: one for particle size analysis and the other for interstitial water and sediment chemistry analysis. Upon retrieval, sediment samples were sealed and stored vertically at 4° C in the dark. Samples were then transferred to a field laboratory for processing.

180. Field processing of the samples for particle size analysis consisted of removing the top 10 cm from one of the two cores collected at each station. Each 10-cm section was gently mixed for several

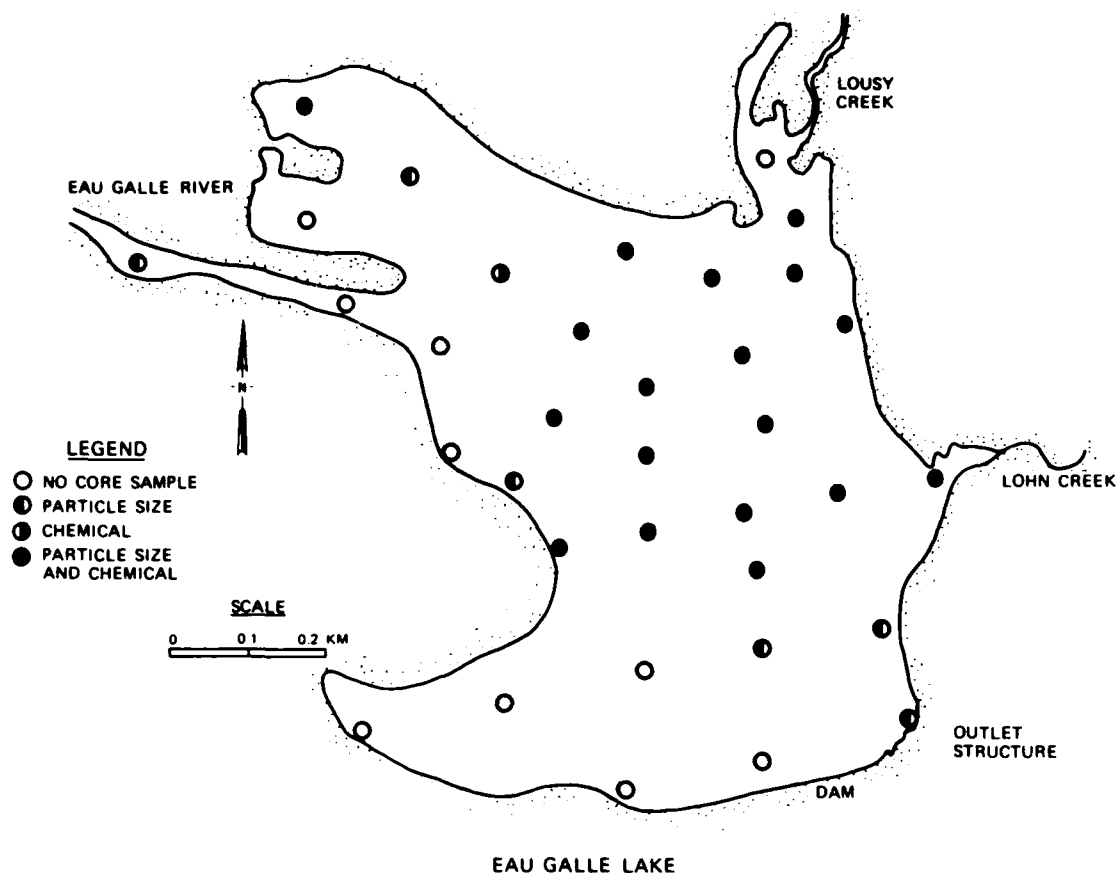


Figure 36. Eau Galle Lake sampling stations indicating type of sample obtained during sediment study conducted 1-2 February 1980

minutes, placed in 35-ml plastic vials, sealed, and shipped on ice by air-express to the US Department of Agriculture Southwest Watershed Research Center in Tucson, Ariz. Particle size analyses were performed using a MicroTrac Particle Size Analyzer (Leeds and Northrop, North Wales, Pa.). The MicroTrac measures 13 particle size fractions between 1.9 and 176 μ .

181. Sediment cores for interstitial water and sediment analyses were processed in a nitrogen atmosphere. The top 10-cm section of each core was centrifuged for 10 min at 10,000 rpm. Each interstitial water supernatant was decanted and filtered through a prewashed, 0.4- μ Nucleopore polycarbonate filter (Nucleopore Corp., Pleasanton, Calif.). Each filtrate was divided and stored in two clean polyethylene bottles,

of which one was acidified with 50 μ l of concentrated hydrochloric acid. Sediment pellets were stored in Whirl-Pak bags (Nasco, Inc., Fort Atkinson, Wis.) and immediately frozen. Interstitial water and sediment samples were shipped on ice and dry ice, respectively, by air to the USAE, Cold Regions Research and Engineering Laboratory in Hanover, N. H., for chemical analysis. Analyses of nonacidified interstitial water samples were initiated as soon as possible. Chemical analyses for all nitrogen and phosphorus forms (interstitial water and sediment) were determined colorimetrically using a Technicon Autoanalyzer II (Technicon Instruments Corp., Atlanta, Ga.). All organic and inorganic carbon analyses were performed on an OIC Carbon Analyzer (Oceanography International Corp., College Station, Tex.) with a Horiba PIR-2000 infrared detector. Determination of all total iron and manganese was by atomic absorption spectroscopy using a Perkin-Elmer Model 403 AA and a HGA-2200 Heated Graphite Atomizer (Perkin-Elmer Corp., Norwalk, Conn.). Moisture content was determined by weighing and drying the sample for 20 hr at 108° C, and then reweighing and drying for an additional hour at 108° C. Moisture content was calculated as weight loss expressed as a percent of sediment dry weight. More detailed discussions of field sampling, field preparation, and analytical methods can be found in Gunkel et al. (1984) and Jenkins et al. (1981).

Results and Discussion

182. The lakelike morphology of Eau Galle Lake influences sedimentary conditions and promotes sediment focusing. Figure 37 indicates that sediment median particle size is related to depth, with smaller median sizes occurring at depths greater than 3.5 m. A depth of 3.5 m also approximates the depth of mixing during summer stratification. According to Hakanson (1977) particle size distribution is a function of the local energy environment. Littoral areas and areas of turbulent inflow are considered high-energy environments, while the deep, more quiescent basin of the lake is a low-energy environment. Therefore, a

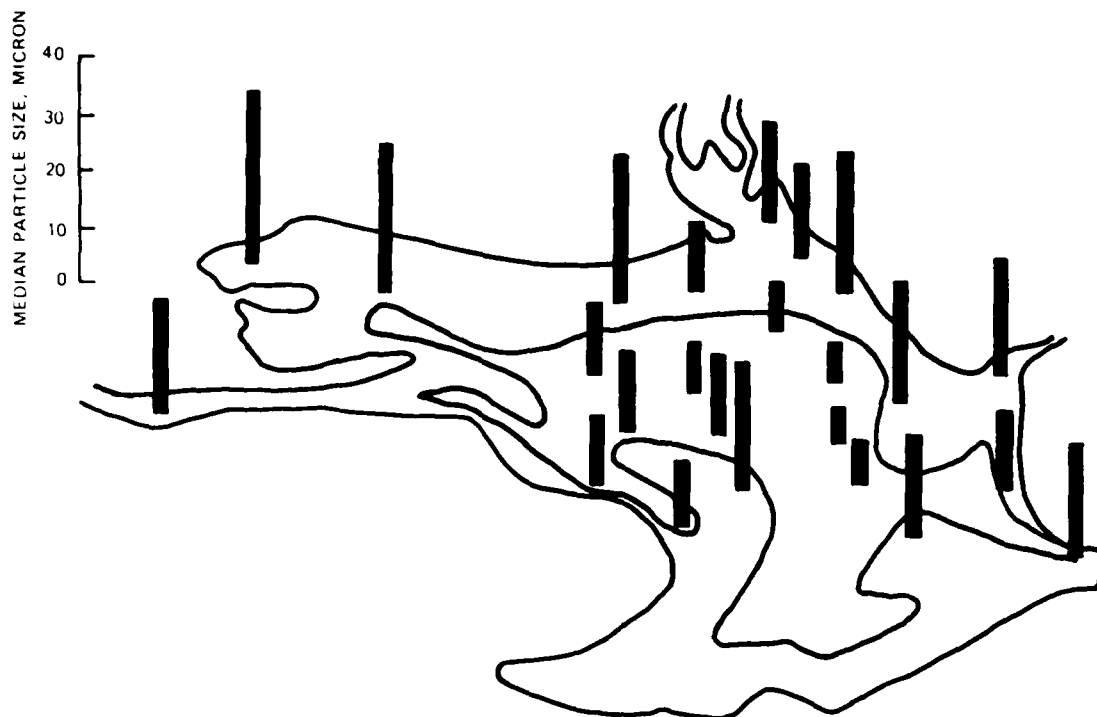


Figure 37. Sediment median particle size represented by bar height for Eau Galle Lake stations. The 3.5-m contour line is shown within the lake

major determinant of the energy environment in this and similar lakes should be depth.

183. Based on the above observation and the absence of any significant correlation between median particle size and distance from the Eau Galle River, the data were subset for further analysis by:

(a) shallow sediments (≤ 3.5 m) and (b) deep sediments (> 3.5 m). The characteristically turbulent nature of the shallow (high-energy) areas is reflected by the relatively uniform distribution of particle volume among each of the 13 particle size classes (Figure 38). The particle size distribution for deep sediments is skewed toward the smaller size classes, suggesting preferential deposition of small particles in the deep areas of the reservoir. This sorting of particles could be the result of differential transport of allochthonous inputs or material resuspension by mixing. Median particle size for deep sediments is

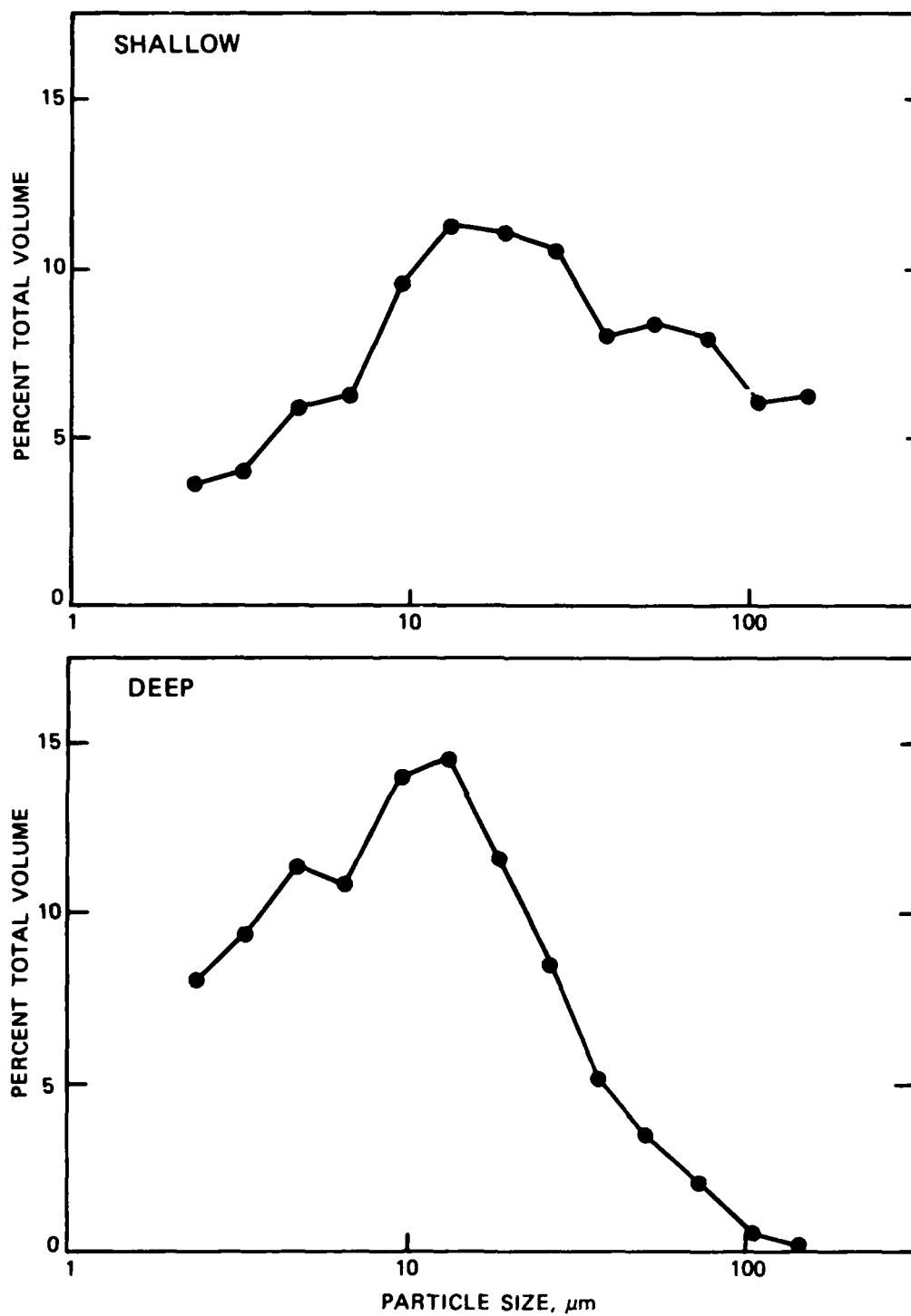


Figure 38. Relationship of percent total volume and particle size for shallow and deep sediments for Eau Galle Lake

significantly ($p < 0.001$) smaller than that of the shallow sediments (10.41 and 21.01 μ , respectively).

184. Sediment moisture content also appears to be depth related. Hakanson (1977) observed that moisture content is inversely related to particle size. Consistent with Hakanson's (1977) findings, Eau Galle deep sediments have a mean moisture content of 67 percent, which is significantly different ($p < 0.001$) from the mean moisture content (45 percent) for shallow sediments. This suggests that the deep areas of Eau Galle are zones of accumulation, while the shallow sediments are subjected to erosion and transport.

185. In addition, Hakanson (1977) reports that nutrient and metal concentrations are associated with and vary proportionally to moisture content. In the deep basins of Lake Vanern, he found enrichments of 150 to 550 percent for organic matter, nitrogen, and phosphorus. All Eau Galle sediment concentrations exhibit significant differences between shallow and deep sediments (Table 11). Concentrations of total organic carbon, nitrogen, phosphorus, iron, and manganese are approximately 1.5 to 2.0 times higher in the deeper sediments than those in shallower sediments. These high concentrations are possibly related to the accumulation of fine particulate matter in the deep basin. A major portion of fine particulate matter consists of phytoplankton, which act as concentrators of epilimnetic carbon, nitrogen, and phosphorus. A significant fraction of their cellular carbon, nitrogen, and phosphorus will be deposited with sedimenting fine particulate matter, thereby enriching the deep sediments. Wetzel (1975) found that iron and manganese bound in the biomass of phytoplankton are not lost in the initial stages of decomposition but rather move with sedimenting organic detritus. Other mechanism for removing dissolved nutrients and metals from the water column would be by the settling out of clays and other fine inorganic particulates.

186. In contrast to the other sediment variables, only inorganic carbon has a higher mean concentration in shallow sediments. This is probably the result of precipitation and deposition of CaCO_3 . In littoral areas, CaCO_3 precipitation is induced when photosynthesis by

Table 11
Eau Galle Lake Mean Values for Shallow and Deep Sediments

Variable	Shallow*	Deep*	p**
Interstitial chemical composition, mg/l			
Soluble reactive phosphorus	0.19	0.30	0.05
Total phosphorus	0.19	0.33	0.05
Total iron	8.42	20.18	0.05
Total manganese	4.97	10.13	0.05
Nitrate nitrite nitrogen	0.01	0.02	NS†
Ammonium nitrogen	10.92	13.43	NS
Total nitrogen	11.57	15.02	NS
Total inorganic carbon	84.40	80.56	NS
Total organic carbon	12.29	16.63	NS
Sediment chemical composition, mg/g			
Total inorganic carbon	7.54	3.33	0.001
Total organic carbon	15.34	30.27	0.005
Total nitrogen	2.03	3.14	0.001
Total phosphorus	0.72	1.35	0.001
Total iron	18.76	31.52	0.001
Total manganese	0.76	1.09	0.01

* Number of observations on which calculations are based: for shallow, n = 10; for deep, n = 9.

** Probability that means are equal.

† Nonsignificant difference ($p > 0.05$).

macrophytes and phytoplankton utilizes CO_2 . Although this precipitation can occur in open waters as well as the littoral areas, little, if any, CaCO_3 will be deposited in deep sediments because of hypolimnetic CO_2 . Carbon dioxide content increases with depth and in the hypolimnion where CO_2 is often abundant, CaCO_3 is dissolved, and bicarbonate content increases. This mechanism effectively prevents enrichment of deeper sediments with inorganic carbon.

187. As a result of accumulation in deep sediments, high interstitial water concentrations of total and soluble reactive phosphorus, iron, and manganese were observed. Since these same variables exhibit enrichment in the sediment phase, it can be expected that dissolved/solid phase interchanges would increase interstitial concentrations. In addition, higher total organic carbon in the sediment fraction should act to increase the rate and intensity of reduction in anaerobic systems (Gunnison and Brannon 1981) and result in higher concentrations of soluble products. The lack of significant enrichment by any of the interstitial nitrogen forms may be the result of their relative ease of mobilization from the sediments (Wetzel 1975).

188. Correlations between particle size classes for deep sediments and various chemical and physical concentrations are presented in Table 12. Percent volume in size classes 2.4 through 4.7 μ are positively correlated with both sediment and interstitial nitrogen (all forms), iron and manganese, sediment total phosphorus, and interstitial carbon. These correlations suggest that particles in the size range of 2.4 through 4.7 μ dominate chemical and biological activities in deep sediments. This results from the fact that particles in this size range contribute 61 percent of the deep sediment's total surface area, while constituting only 29 percent of the total volume. Sly (1977) stressed that the surface area of clay-sized particles is on the order of square meters per gram whereas the surface area of sand grains is only on the order of square centimeters per gram. Fine-grained materials have the greatest potential for chemical and biological interaction because of the importance of surface reactions in sediments (Jones and Bowser 1977).

Table 12

Significant ($p < 0.05$) Correlation Coefficients for the Fau (Galle Deep Sediments)
(Particle Size Values Are Midpoints of Size Ranges)

Variable	Particle Size, μ												
	150	106	75	53	38	27	19	13	9.4	6.6	4.7	3.3	2.4
Interstitial chemical composition													
Total inorganic carbon	-0.44	NS*	-0.81	-0.47	NS	-0.46	-0.44	-0.58	NS	NS	NS	0.58	0.61
Total organic carbon	NS	NS	-0.82	-0.69	NS	-0.68	-0.68	-0.53	NS	NS	0.61	0.81	0.80
Nitrate nitrite nitrogen	NS	-0.44	-0.45	-0.67	-0.45	NS	-0.70	NS	0.48	0.66	0.64	0.59	0.57
Ammonium nitrogen	NS	NS	-0.79	-0.45	NS	-0.54	-0.55	-0.59	NS	NS	0.48	0.63	0.61
Total nitrogen	NS	NS	-0.78	-0.48	NS	-0.56	-0.56	-0.57	NS	NS	0.51	0.66	0.63
Soluble reactive phosphorus	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
Total phosphorus	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
Total iron	NS	NS	-0.78	-0.52	NS	-0.67	-0.54	-0.49	0.46	NS	0.58	0.71	0.66
Total manganese	NS	NS	-0.76	NS	NS	-0.50	-0.54	-0.63	NS	NS	NS	0.59	0.59
Sediment chemical composition													
Total inorganic carbon	NS	NS	NS	NS	NS	NS	NS	0.56	NS	NS	NS	NS	NS
Total organic carbon	NS	NS	NS	NS	NS	NS	0.55	0.67	NS	NS	NS	NS	NS
Total nitrogen	NS	NS	-0.69	NS	NS	-0.68	-0.53	NS	NS	NS	0.53	0.63	0.56
Total phosphorus	NS	NS	-0.85	-0.60	NS	-0.71	-0.66	-0.67	NS	NS	0.61	0.59	0.57
Total iron	NS	NS	-0.45	NS	NS	-0.47	NS	NS	NS	NS	NS	0.57	0.45
Total manganese	NS	NS	-0.63	NS	NS	NS	NS	-0.57	NS	NS	NS	0.43	0.45
Median particle size	NS	0.80	0.52	0.87	0.84	0.85	0.73	NS	-0.80	-0.88	-0.97	-0.91	-0.87
Column depth	NS	NS	NS	NS	NS	-0.45	-0.63	NS	0.44	0.48	0.57	0.54	0.54

* NS = correlation was not significant.

189. Sediment organic carbon is correlated with percent volumes in the 13- and 19- μ size classes. Since organic carbon is usually correlated with clays (Thomas 1969), the correlation observed here with larger than clay-sized particles suggests the existence of detrital particulate organic matter rather than an organic film on an inorganic particle.

190. There is a general lack of significant correlation between chemical composition and percent volume in the 6.6-, 9.4-, 106-, and 150- μ size classes. The coefficients of variation are lowest for the 6.6- and 9.4- μ size classes ($CV = 12$ and 8 , respectively). In addition to having the lowest CV 's, these two size classes account for 25 percent of the deep sediment's total volume, which suggests that these two size classes are relatively evenly distributed across the deep sediment and would therefore exhibit little relation to chemical composition. The high variability observed in the 106- and 150- μ size classes ($CV = 149$ and 316 , respectively), and their small contribution (0.7 percent) to total volume of deep sediment, is probably the reason for the lack of correlation.

191. Shallow sediment exhibits a general lack of correlation between particle size and chemical composition. This lack of correlation may imply that the variability of littoral sediment chemical composition may be a function of localized influences (i.e., inflows, macrophytes, and direct runoff) rather than particle size. Turbulence in the littoral zone has reduced particle size variability between stations, thereby reducing correlations between particle size and chemical composition.

Conclusions

192. Sedimentary conditions in Eau Galle are primarily a function of basin morphology (i.e., depth). The combination of Eau Galle's circular shape, multiple inflows, and deep central basin results in sediment deposition by focusing. In this regard, two sedimentary environments can be distinguished within the reservoir. A high-energy

environment, which is dominated by turbulent processes (e.g., flow, pool fluctuation, wind, etc.), comprises the littoral and inflow areas of the reservoir. The turbulent nature of this environment discourages the permanent deposition of fine particulates. Sediments in these areas are characterized as having a larger median particle size, low moisture content, and lower nutrient, metal, and organic matter concentrations.

193. Conditions in the low-energy environment (i.e., deep portions of the lake) are less turbulent and provide an area for sediment accumulation. Sediments in this area are characteristically higher in moisture content and are comprised of relatively smaller particles. Higher concentrations of nutrients, metals, and organic matter are also found in these deeper sediments. These characteristics, along with expected exchanges between the sediment and overlying water, infer that deep sediments are likely to influence reservoir water quality.

References

- Davis, M. B. 1973. Redeposition of pollen grains in lake sediments. *Limnology and Oceanography* 18:44-52.
- Davis, M. B., and L. B. Brubaker. 1973. Differential sedimentation of pollen grains in lakes. *Limnology and Oceanography* 18:635-646.
- Gunkel, R. C., Jr., R. F. Gaugush, R. H. Kennedy, G. E. Saul, J. H. Carroll, and J. E. Gauthey. 1984. A comparative study of sediment quality in four reservoirs. Technical Report E-84-2. US Army Engineer Waterways Experiment Station, Vicksburg, Miss.
- Gunnison, D., and J. M. Brannon. 1981. Characterization of anaerobic chemical processes in reservoirs: problem description and conceptual model formulation. Technical Report E-81-6. US Army Engineer Waterways Experiment Station, Vicksburg, Miss.
- Hakanson, L. 1977. The influence of wind, fetch, and water depth on the distribution of sediments in Lake Vanern, Sweden. *Canadian Journal of Earth Science* 14.
- Jenkins, T. F., P. W. Schumacher, D. C. Leggett, J. H. Cragin, and C. L. Grant. 1981. Chemical analysis of sediment and interstitial waters from selected Corps reservoirs. ODMR No. WESRF 80-164. US Army Engineer Cold Regions Research and Engineering Laboratory, Hanover, N. H. 29 pp.

Jones, B. F., and C. J. Bowser. 1977. The mineralogy and related chemistry of lake sediments. In A. Lerman, ed. Lake Chemistry, geology, physics. Springer-Verlag, New York. 363 pp.

Sly, P. G. 1977. Sedimentary processes in lakes. In A. Lerman, ed. Lake chemistry, geology, physics. Springer-Verlag, New York. 363 pp.

Thomas, R. L. 1969. A note on the relationship of grain size clay content, quartz, and organic carbon in some Lake Erie and Lake Ontario sediments. Journal of Sedimentary Petrology 39:803-809.

Wetzel, R. G. 1975. Limnology. W. B. Sanders Co., Philadelphia, Pa. 743 pp.

Wetzel, R. G., P. H. Rich, M. C. Miller, and H. I. Allen. 1972. Metabolism of dissolved and particulate detrital carbon in a temperate hard-water lake. Memorie Dell' Instituto Italiano Di Idrobiologia 29 (Suppl.):185-243.

PART IX: HYPOLIMNETIC PHOSPHORUS DYNAMICS*

Introduction

194. Lakes experiencing anoxia during summer stratification often exhibit marked increases in hypolimnetic phosphorus concentrations, primarily due to phosphorus release from sediment (Mortimer 1971). This phosphorus may become a significant source of phosphorus for the epilimnion via vertical transport, particularly when external loading is minimal. The vertical transport of hypolimnetic phosphorus is thought to be governed both by turbulent diffusion and weather-related mixing processes. Typically, hypolimnetic phosphorus concentrations increase substantially with depth and through time, suggesting an upward flux by turbulent diffusion (Imboden and Emerson 1978, Wodka et al. 1983). Weather-induced mixing events can then entrain phosphorus-rich water to the epilimnion. For instance, Stauffer and Lee (1973), Larson, Schultz, and Malueg (1981), Kortmann et al. (1982), and Stauffer and Armstrong (1984), reported that mixing induced by wind stress resulted in the downward movement of the thermocline with accompanying increases in epilimnetic phosphorus concentrations. These increases were followed by the occurrence of algal blooms.

195. Gaugush (1986) has distinguished several factors which make Eau Galle Lake susceptible to weather-induced mixing events during summer stratification. First, a shallow morphometry results in a high center of gravity (i.e., near the 3-m depth). Movement of the thermocline past this depth acts to destabilize the thermal structure of the lake. Second, summer releases occur primarily through a hypolimnetic release gate (7 m deep) causing a gradual increase in hypolimnetic temperature; this in turn reduces density differences in the metalimnion. These two conditions act to reduce thermal stability and make the lake susceptible to mixing during wind events.

* Part IX was written by William F. James, Robert H. Kennedy, and Robert F. Gaugush.

196. Gaugush (1986) identified two types of mixing events in Eau Galle Lake and described their impact on epilimnetic phosphorus concentrations. The first event is a small-scale mix which results in the loading of phosphorus to the epilimnion and a concomitant increase in algae. The second is a large-scale mix that disrupts the metalimnetic barrier, which causes the reintroduction of dissolved oxygen to the previously anoxic hypolimnion and a net loss of nutrients and metals from the water column.

197. This study was conducted to determine the impact of large-scale mixing events on hypolimnetic phosphorus dynamics in Eau Galle Lake. Specific objectives were to determine rates of hypolimnetic phosphorus accumulation and possible exchanges between the epilimnion and hypolimnion during these mixing events.

Methods

198. Eau Galle Lake, inflowing streams, and the outflow were sampled biweekly during the summer of 1982 for total phosphorus (TP) and soluble reactive phosphorus (SRP). Water samples were collected in acid-washed, polyethylene bottles at station 20 at 1-m intervals from the surface to 0.5 m from the sediment surface. Samples were analyzed for TP and SRP following standard preparation and digestion procedures (American Public Health Association (APHA) 1980) on a technicon autoanalyzer. SRP was determined for 0.45- μ filtered samples. Anoxic samples were pumped directly into a 50-cc plastic syringe to prevent air exposure, then pushed through a membrane for SRP determination. Temperature and dissolved oxygen measurements were determined at the same stations and depth intervals with a Hydrolab Surveyor calibrated against an NBS thermometer and Winkler titrations (APHA 1980).

199. During the establishment of thermal stratification and anoxia (May-July), hypolimnetic samples were collected at 10-cm intervals with a syringe sampler as described by Blakar (1979). The sampler consisted of copper tubing manifolds with 24 ports to which were attached syringes (Figure 39). Attached to each syringe was a Swinnex

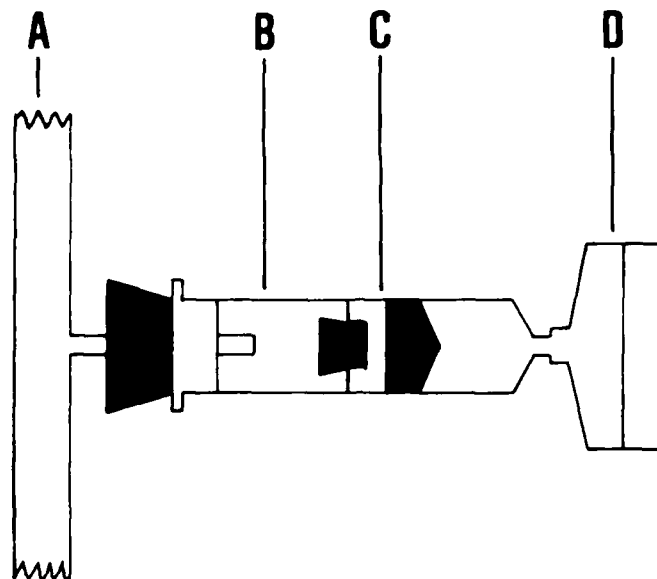


Figure 39. Detailed diagram of syringe sampler showing the copper manifold with sampling port (A), the 50-cc syringe (B), the moving plunger with rubber stopper (C), and the filter holder (D)

filter holder containing an acid-washed 0.45- μ filter. A gauged pressure of 20 psi (138 kPa) was maintained in the device during deployment with a tire pump to prevent premature sampling. The sampling device, which was placed on a pulley-type suspension system located in the deepest part of the lake (9 m), was carefully lowered to the bottom, then left undisturbed for 2 hr. Near-bottom filtered samples were collected by releasing the pressure, then maintaining a vacuum of 20 psi with a hand pump before retrieval. Total collection time was 3 to 4 hr.

200. Water samples were then processed immediately and analyzed for SRP the same day. In addition, near-bottom filtered samples were analyzed for iron by atomic absorption spectrophotometry using standard procedures (APHA 1980). Close-interval samples were collected frequently in May (3- to 5-day intervals) and at 1- to 3-week intervals in June and July. In situ measurements of temperature and dissolved oxygen were made at the same depth intervals.

201. The metalimnetic zone was determined by calculating the relative thermal resistance to mixing (RTRM) statistic according to the equation

$$RTRM_z = (P_z - P_{z+1}) / (P_{5^\circ C} - P_{4^\circ C})$$

where P = density of water at depth z . An RTRM value of 30 was used to define the upper and lower metalimnetic boundaries (Kortmann et al. 1982, Gaugush 1986).

202. Deposition rates were measured at biweekly intervals with sedimentation traps suspended at the 4-m (metalimnetic zone) and 8-m (1 m from lake bottom) depths. The sedimentation traps and analytical procedures are described in James (1986).

Results and Discussion

203. Seasonal and vertical patterns in temperature were pronounced from May to late July (Figure 40). Following ice-out and vernal turnover in April, the lake exhibited stratified conditions in early May. Estimates of the metalimnetic zone were determined by comparing temperature profiles with the relative thermal resistance to mixing (RTRM = 30) statistic. In May, the metalimnion covered a large vertical area (from 2 to 5 m of depth) and closely reflected vertical gradients in temperature. During the passage of a cold front and high wind activity in June, the metalimnetic zone descended sharply from a mean 3.5 m to 6.5 m, indicating the occurrence of a large-scale mixing event. In July, the metalimnetic zone ascended to a mean depth of 4 m and again covered an expansive area.

204. Patterns in dissolved oxygen closely corresponded to temperature variations and vertical changes in the metalimnetic zone (Figure 41). Anoxia had developed below 6.5 m in late May, and dissolved oxygen was less than 5.0 mg/l below 3.5 m. During the mixing event in June, the depth of the top of the anoxic zone descended to 7.0 m, and pronounced increases in dissolved oxygen occurred to a depth of 6.0 m.

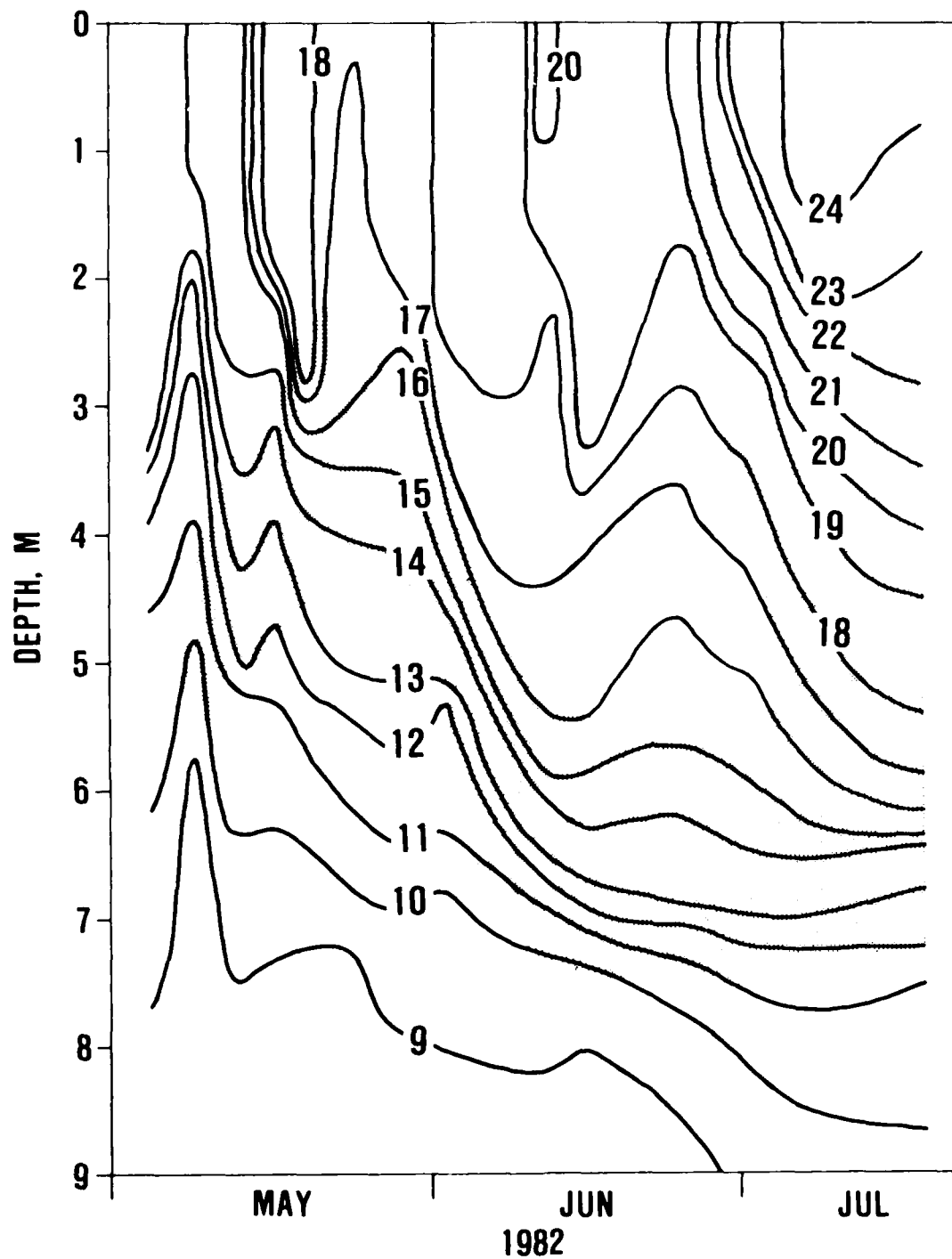


Figure 40. Temperature ($^{\circ}\text{C}$) at station 20 from May to July 1982.
Shaded area represents approximate area of metalimnion

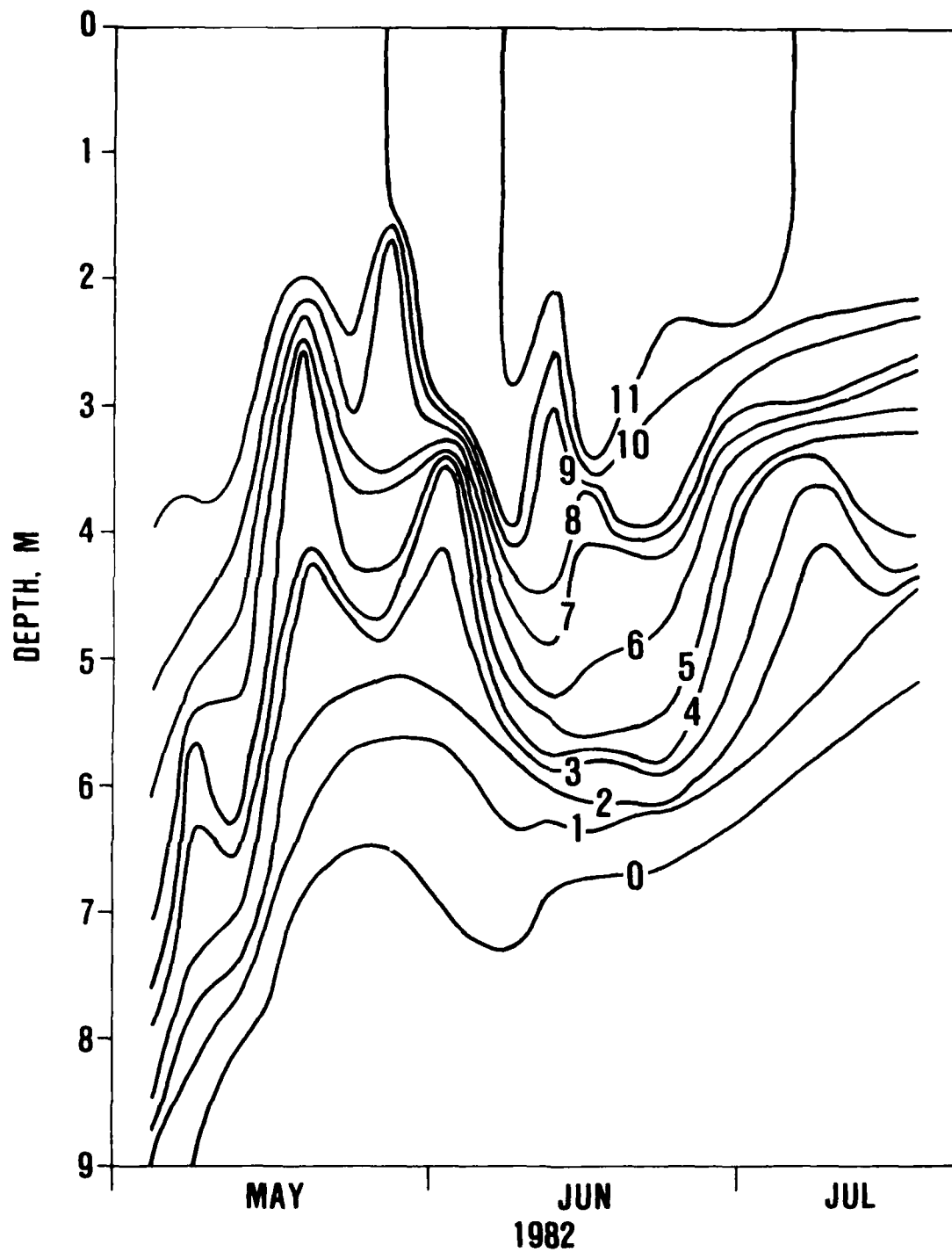


Figure 41. Dissolved oxygen concentrations (mg/l) at station 20 from May to July 1982

The zone of anoxia then gradually increased, and low dissolved oxygen conditions had developed at the 5.0-m depth in late July.

205. Hypolimnetic SRP dynamics, as identified by close-interval sampling, were clearly influenced by diffusional processes and the mixing event in June (Figure 42). Accumulation of SRP near the sediment surface began in May coincident with the beginning of anoxia, and marked vertical gradients were evident in late May. During the June mixing event, SRP decreased substantially between the 4.5- and 6.5-m depths, and distinct gradients in SRP were confined to a small (20-cm) vertical zone located near the lower metalimnetic boundary. From June through mid-July, SRP continued to increase near the sediment surface.

206. The origin of hypolimnetic SRP was clearly the sediment surface (Figure 43). Concentrations of SRP were highest at the sediment/water interface, and gradients of decreasing SRP were apparent toward the lake surface. The SRP at each depth increased with time, suggesting an upward movement. The increases in SRP at the sediment surface closely coincided with the disappearance of dissolved oxygen in the bottom water, and there was a strong relationship between the occurrence of hypolimnetic SRP and filterable iron (DFe) near the sediment surface, suggesting that iron-phosphorus disassociation was the mechanism for SRP release (Figure 44). However, as will be discussed, recently deposited seston may have been an additional source of phosphorus to the hypolimnion.

207. From May until mid-July, the extent of SRP accumulation was confined to the region near the lower metalimnetic boundary, and vertical SRP gradients appeared to be influenced by the hypolimnetic withdrawal structure located at the 7-m depth (Figure 43). In May, when the hypolimnetic volume was large (0.61 km^3), SRP accumulation was evident between the 4.5- and 9-m depths. Accumulation through time was greatest between the sediment surface and the 7-m depth, and thus below the hypolimnetic withdrawal structure. Accumulation of SRP above the structure (i.e., above the 7.0-m depth) was noticeable and accounted for 24 percent of the hypolimnetic increase in May. Accumulation was not evident above the lower metalimnetic boundary. In June, thermocline migration

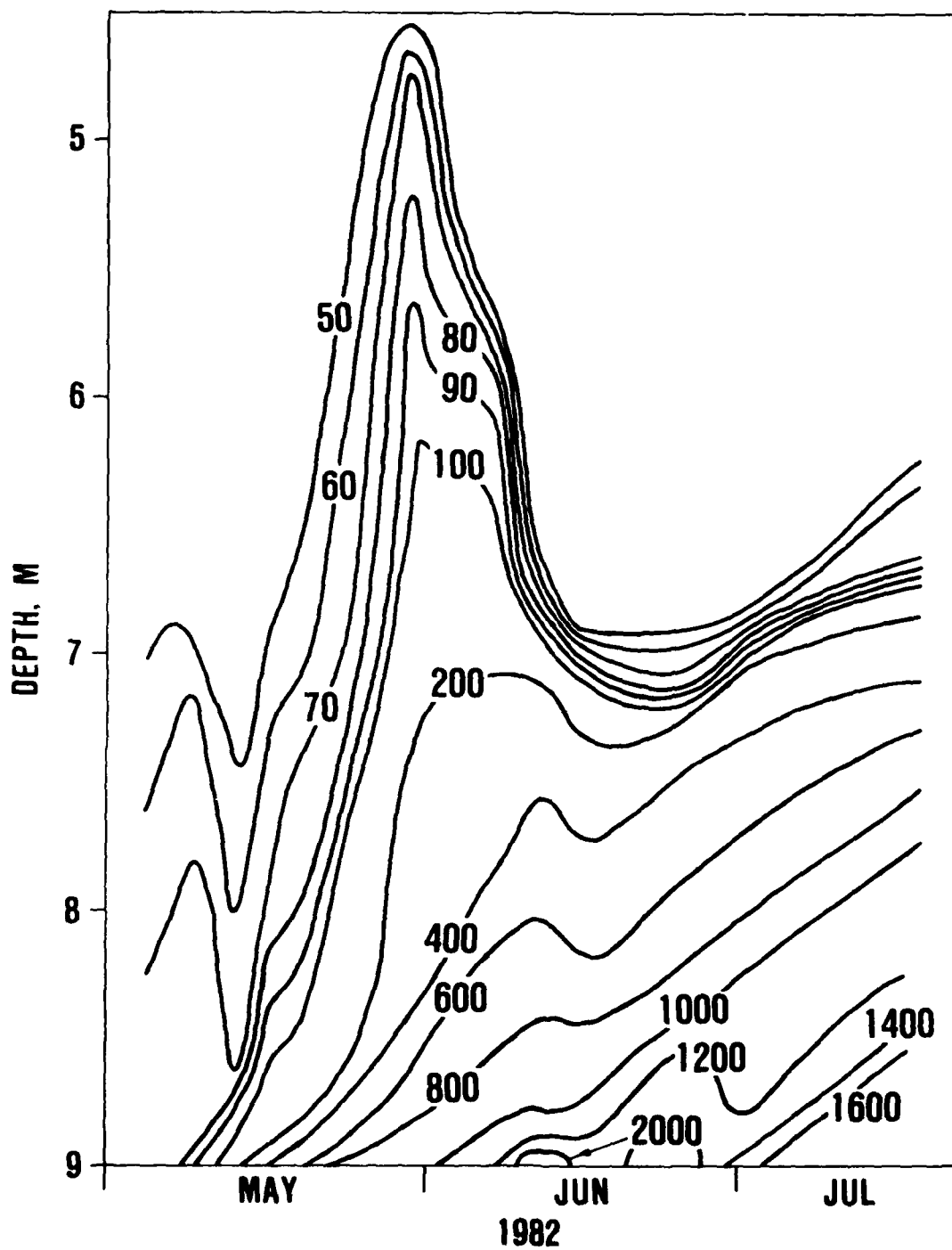


Figure 42. SRP concentrations ($\mu\text{g/l}$) from 4- to 9-m depths at station 20. Note depth scale change

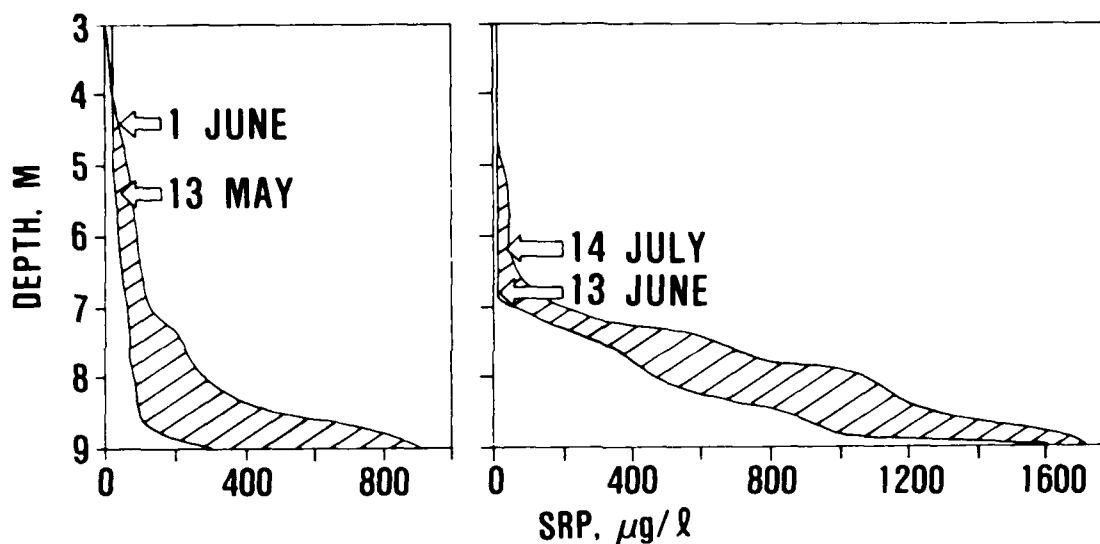


Figure 43. Detailed changes in SRP concentrations ($\mu\text{g}/\ell$) before and during the June mixing event. Striped areas represent zones of SRP accumulation on successive dates. Arrows represent approximate locations of lower metalimnetic boundary

caused a decrease in the hypolimnetic volume to 0.23 km^3 . Marked gradients in SRP concentration were observed below a depth of 6.5 m, and further accumulation was evident between June and mid-July. However, SRP accumulation above 6.5 m was small, with the appearance of SRP in these strata coinciding with a gradual increase in hypolimnetic volume.

208. Changes in the SRP content of the hypolimnion were determined for the period of early summer stratification and compared with inputs and outputs (Figure 45). Hypolimnetic SRP content, which was defined as the mass of SRP (kilograms) below a depth of 4.5 m, increased rapidly from early-May to June. During and following the June mixing event, SRP continued to increase, although at a slower rate. SRP loading and releases were high in May, and decreased to minimal levels from June through July.

209. Changes in the TP and SRP content of the epilimnetic and metalimnetic zone (0 to 4.5 m) indicate that the transfer of phosphorus from the hypolimnion to the epilimnion was probably minimal before and during the mixing event (Table 13). Before the mixing event, SRP increased slightly to 41 kg in late May. During the mixing event in

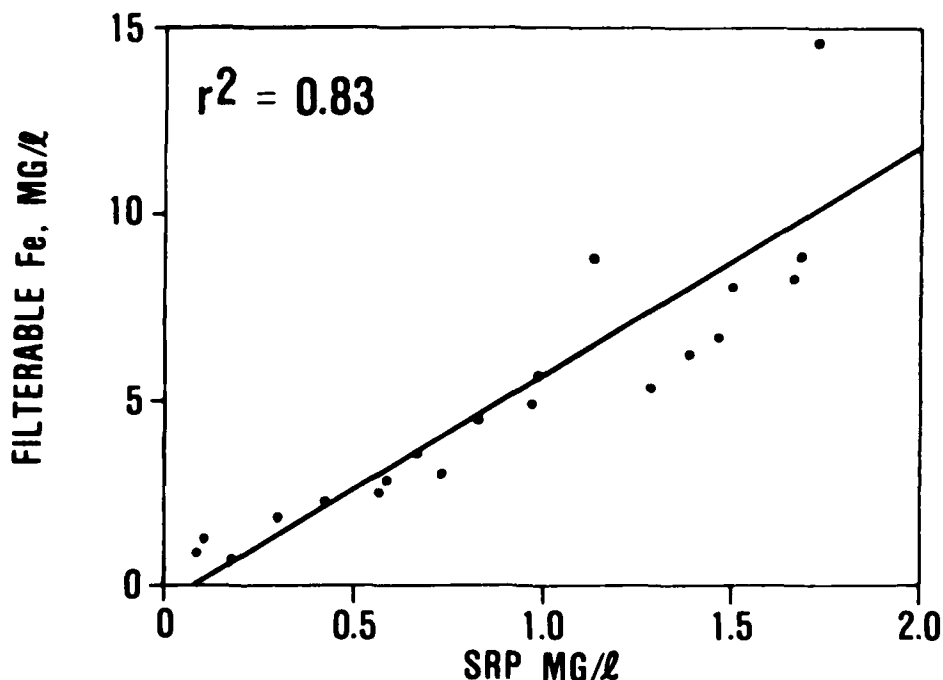


Figure 44. Relationship between SRP and DFe near the sediment surface

June, SRP decreased to 11 kg, then increased slightly, remaining constant until the end of June. TP mass displayed a maximum peak of 167 kg in early May, which was associated with elevated chlorophyll a levels and the occurrence of a diatom bloom (Barko et al. 1986), indicating that phosphorus was incorporated into the algal fraction. The peak occurred shortly after a period of snowmelt runoff, suggesting that sources of phosphorus for algae were external rather than internal. TP descended in late May, coincident with a decrease in diatom abundance in the water column. In June, TP continued to decrease during the mixing event, indicating that upward transfer of phosphorus and incorporation into a particulate fraction did not occur. Rather, it appeared that phosphorus was lost from the epilimnion by deposition.

210. Thus, in early June there was an apparent loss of SRP between the 4.5- and 6.5-m depths which could not be accounted for by an increase in epilimnetic phosphorus mass. However, since the mixing event resulted in a decrease in the anoxic zone and the introduction of

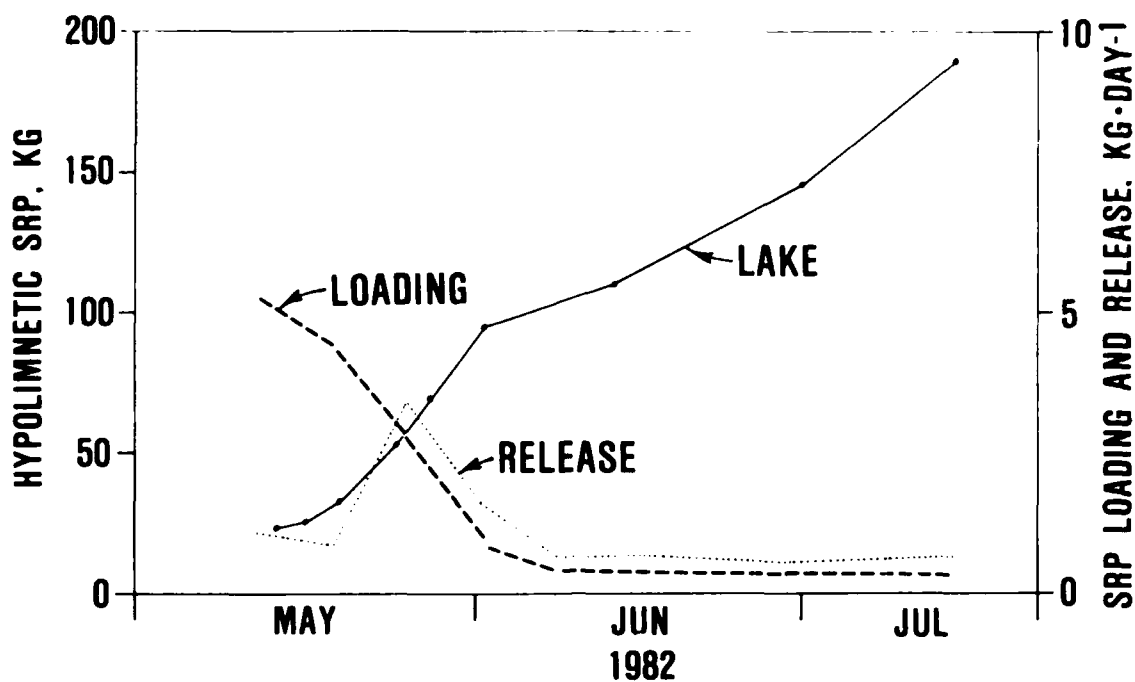


Figure 45. Changes in SRP content (kilograms) of the hypolimnion, SRP loading and releases (kilograms per day)

Table 13
Changes in Epilimnetic TP and SRP Content Determined
Between the Surface and 4.5-m Depth

Date, 1982	TP, kg	SRP, kg	SRP/TP kg
4 May	99	6	0.06
11 May	167	13	0.08
18 May	134	30	0.22
25 May	126	41	0.33
1 June	104	36	0.35
8 June	94	11	0.12
15 June	81	22	0.27
29 June	77	19	0.25

dissolved oxygen greater than 5.0 mg/l to a depth of 5.5 m, loss of phosphorus from the upper hypolimnion might have occurred by deposition with oxidized iron. Deposition measurements made during the period were converted to hypolimnetic depositional gains or losses by subtracting sediment trap rates measured at the 4-m depth (upper metalimnion) from rates measured at the 8-m depth (1 m from bottom). A positive difference represented a hypolimnetic depositional gain, and a negative difference represented a loss (Table 14). The hypolimnion experienced particulate phosphorus and iron gains throughout the study, particularly in May, when the decline of a diatom bloom occurred. However, shortly after the mixing event in June, increased hypolimnetic gains in phosphorus and iron were observed, suggesting that losses in hypolimnetic SRP might be accounted for by deposition rather than entrainment into the epilimnion.

211. Hypolimnetic accumulation rates, corrected for inflows and outflows, were calculated to determine possible effects from the mixing event (Figure 46). Changes in SRP mass between 4.5 and 9.0 m were divided by the lake's surface area (0.6232 km^2) and the time interval to calculate rates. Accumulation rates increased rapidly to a peak of $10 \text{ mg/m}^2/\text{day}$ in late May, when the hypolimnetic volume was large and SRP gradients were detected up to the lower metalimnetic boundary. During the mixing event, a marked decrease in the accumulation rate was observed. This appeared to be consistent with a probable loss of SRP from the upper hypolimnion and a decrease in hypolimnetic volume. Rates then began increasing by late-June to July as the hypolimnetic volume and zone of anoxia increased.

212. In general, it appeared that the large-scale mixing event in June had a pronounced effect on vertical SRP gradients in the water column and rates of hypolimnetic SRP accumulation. Probable mechanisms for these observations were a loss of phosphorus from the upper hypolimnion and a decrease in the hypolimnetic volume.

213. To further support these contentions, a comparison was made with another mixing event which occurred in early-August 1982. As in June, mixing caused a pronounced descent of the thermocline in

Table 14
Deposition of P and Fe ($\text{mg}/\text{m}^2/\text{day}$) in the Hypolimnion

Date, 1982	P $\text{mg}/\text{m}^2/\text{day}$	Fe $\text{mg}/\text{m}^2/\text{day}$
28 Apr-13 May	36.7	147.6
13 May-24 May	13.7	176.6
24 May-11 Jun	14.1	146.6
11 Jun-21 Jun	20.1	210.1
21 Jun-2 Jul	12.6	146.2
2 Jul-19 Jul	3.1	131.0

Note: Rates were calculated as the difference between deposition at the 4- and 8-m depths at station 20.

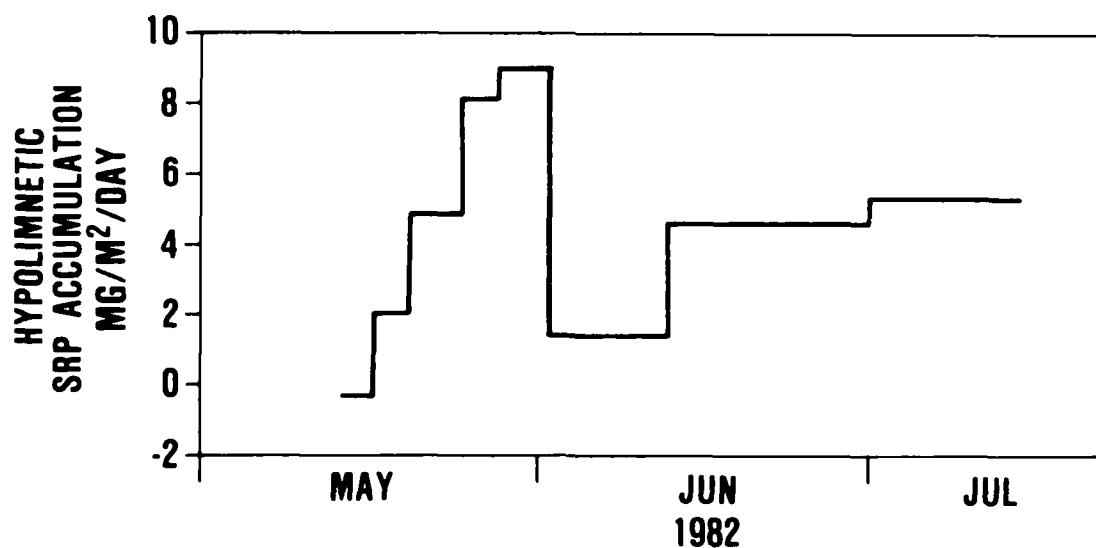


Figure 46. Hypolimnetic SRP accumulation rates (milligrams/square meter/day)

early-August and introduction of dissolved oxygen to the hypolimnion (Figure 47). During both periods of extensive mixing, epilimnetic TP content displayed a net loss or little fluctuation, rather than gains, which might be expected (Figure 48). Hypolimnetic TP content also exhibited a net loss in June; however, a slight increase in TP content was detected during the mixing event in early-August. Marked depositional increases in particulate phosphorus and iron in the hypolimnion coincided with major mixing events in June (as discussed), late-July, and September, during autumnal turnover (Figure 49). These results further indicated that iron and phosphorus were lost from the hypolimnion by deposition during extensive thermocline migration. During periods when major mixing did not occur, hypolimnetic TP mass increased rapidly and particulate gains were lower.

214. Rates of hypolimnetic TP accumulation fluctuated with the extent of anoxia and periods of increased particulate deposition in the hypolimnion (Figure 50). For instance, in May, late-June, and mid-August, hypolimnetic accumulation rates were high when the hypolimnetic volume was large, as suggested by low DO concentrations. Rates of hypolimnetic accumulation decreased substantially in early-June and, to a smaller extent, in early-August, consistent with a decrease in hypolimnetic volume and increased hypolimnetic deposition. These trends were similar to the results obtained from close-interval sampling. The hypolimnetic TP accumulation rate also decreased in July, coincident with a slight depression in hypolimnetic anoxia. However, this was not accompanied by a depositional increase in the hypolimnion. Rather, epilimnetic TP content increased in July (Figure 47) suggesting an upward transfer of phosphorus (Gaugush 1986). The lakewide TP accumulation rate exhibited a pronounced increase during this period, which reflected TP concentration increases in the epilimnion.

215. The release of TP from the hypolimnetic withdrawal structure was responsible for a substantial percentage of the hypolimnetic TP accumulation rate. From late-June through August, output of TP accounted for 20 to 90 percent of the accumulation rate, indicating that a significant amount of the internal load was leaving the lake.

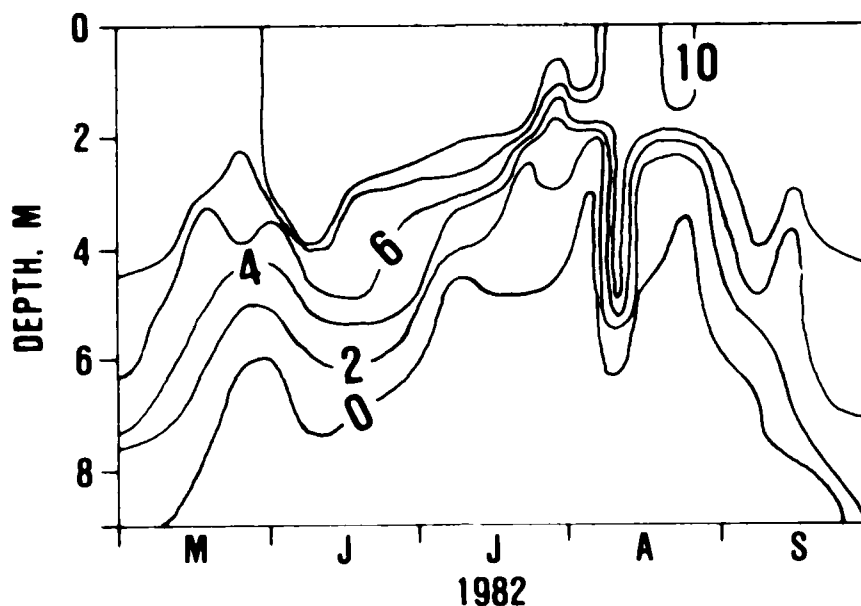


Figure 47. Seasonal patterns in dissolved oxygen (milligrams/liter) at station 20 from May through September 1982

Summary

216. Eau Galle Lake is susceptible to mixing which can disrupt the thermal structure of the water column. The susceptibility to mixing is thought to be governed by the lake's shallow morphometry and a hypolimnetic release, which causes a more rapid heating of this region, thereby reducing thermal stability. The passage of cold fronts and occurrence of high wind activity in summer can affect the extent of thermal instability and mixing in two ways (Gaugush 1986). First, when little or no change in the heat content of the lake occurs during the passage of a cold front, mixing results in only a slight depression of the thermocline. Second, when a significant heat loss occurs in the water column, mixing acts as a short-lived turnover, resulting in a large depression of the thermocline. Hypolimnetic phosphorus dynamics appear to be directly and indirectly affected by these episodic events. Reported here are the results of large-scale mixing events on hypolimnetic phosphorus dynamics.

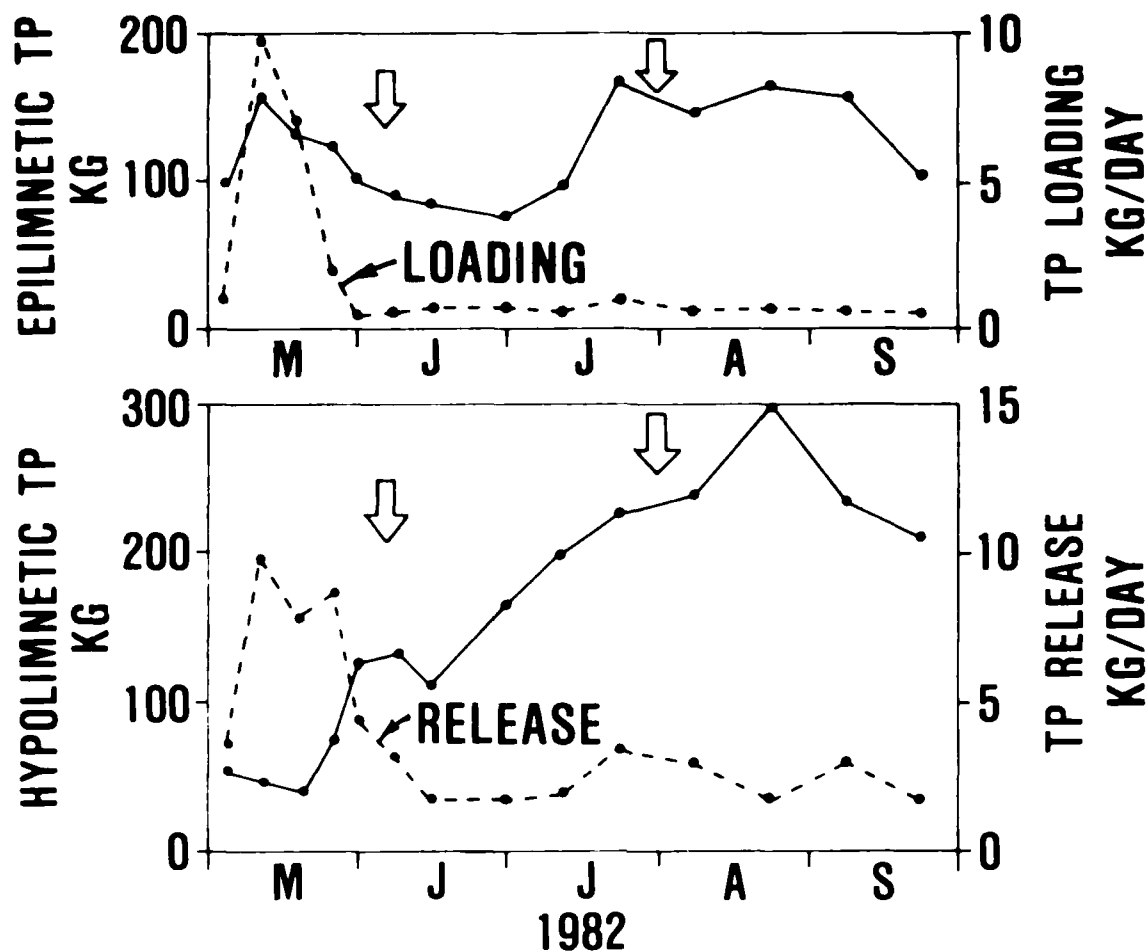


Figure 48. Seasonal changes in epilimnetic and hypolimnetic TP (kilograms), and TP loading and releases (kilograms/day). Arrows denote occurrences of introduction of dissolved oxygen into the hypolimnion

217. Sediment is an important internal source of phosphorus in Eau Galle Lake during summer. Shortly after the onset of thermal stratification, hypolimnetic SRP accumulation was pronounced, particularly near the sediment surface. The accumulation of SRP in this region was apparently linked to release from the sediment surface by disassociation from iron during anoxia. There was a good relationship between the occurrence of anoxia and near-bottom increases in SRP and DFe concentrations. However, recently sedimented material may have also accounted for some of the increase near the bottom. Hypolimnetic depositional

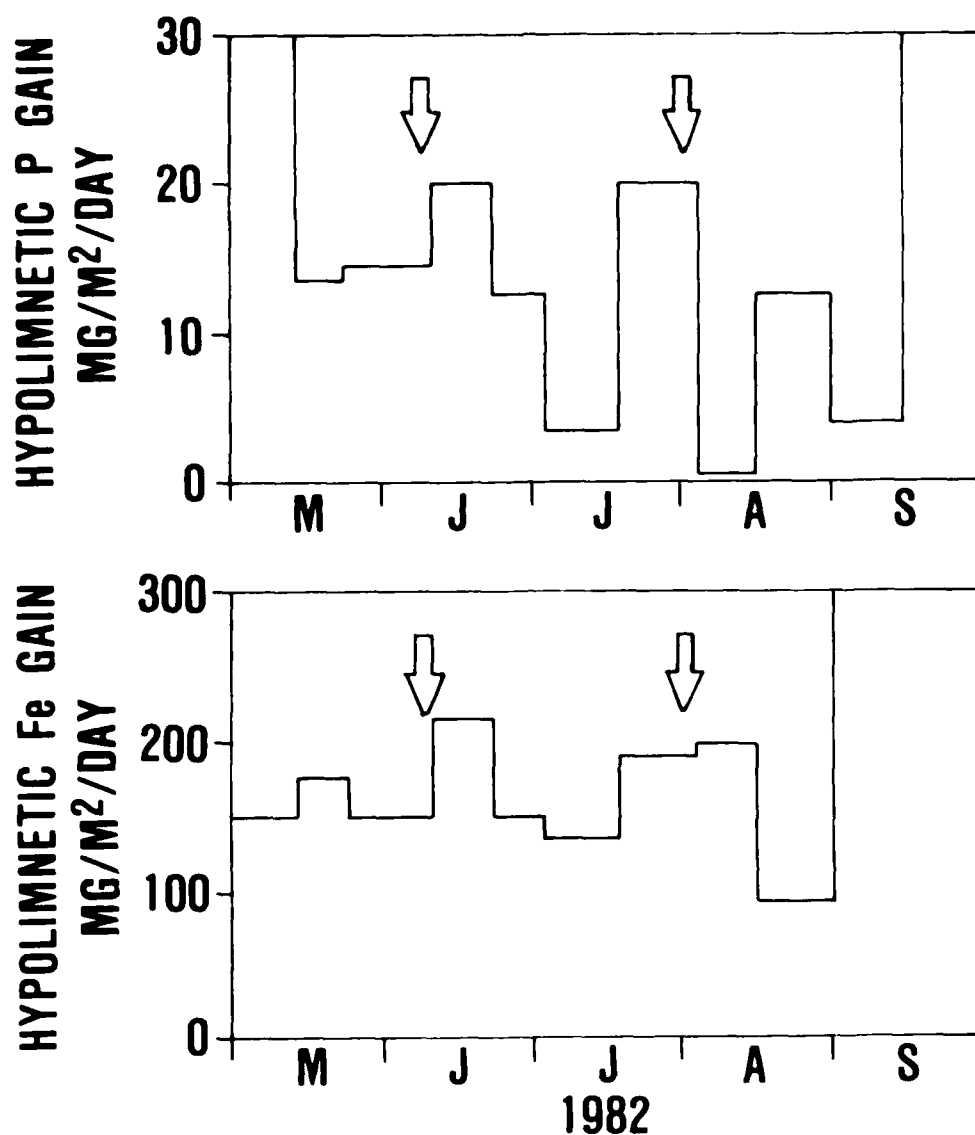


Figure 49. Seasonal changes in hypolimnetic deposition of particulate phosphorus and iron (milligrams/square meter/day)

gains in particulate phosphorus were marked in early May and coincided with an algal crash, suggesting that the settling material consisted of algal remains (James 1986). Leaching and decomposition of algal detritus may have been an important additional source of hypolimnetic phosphorus.

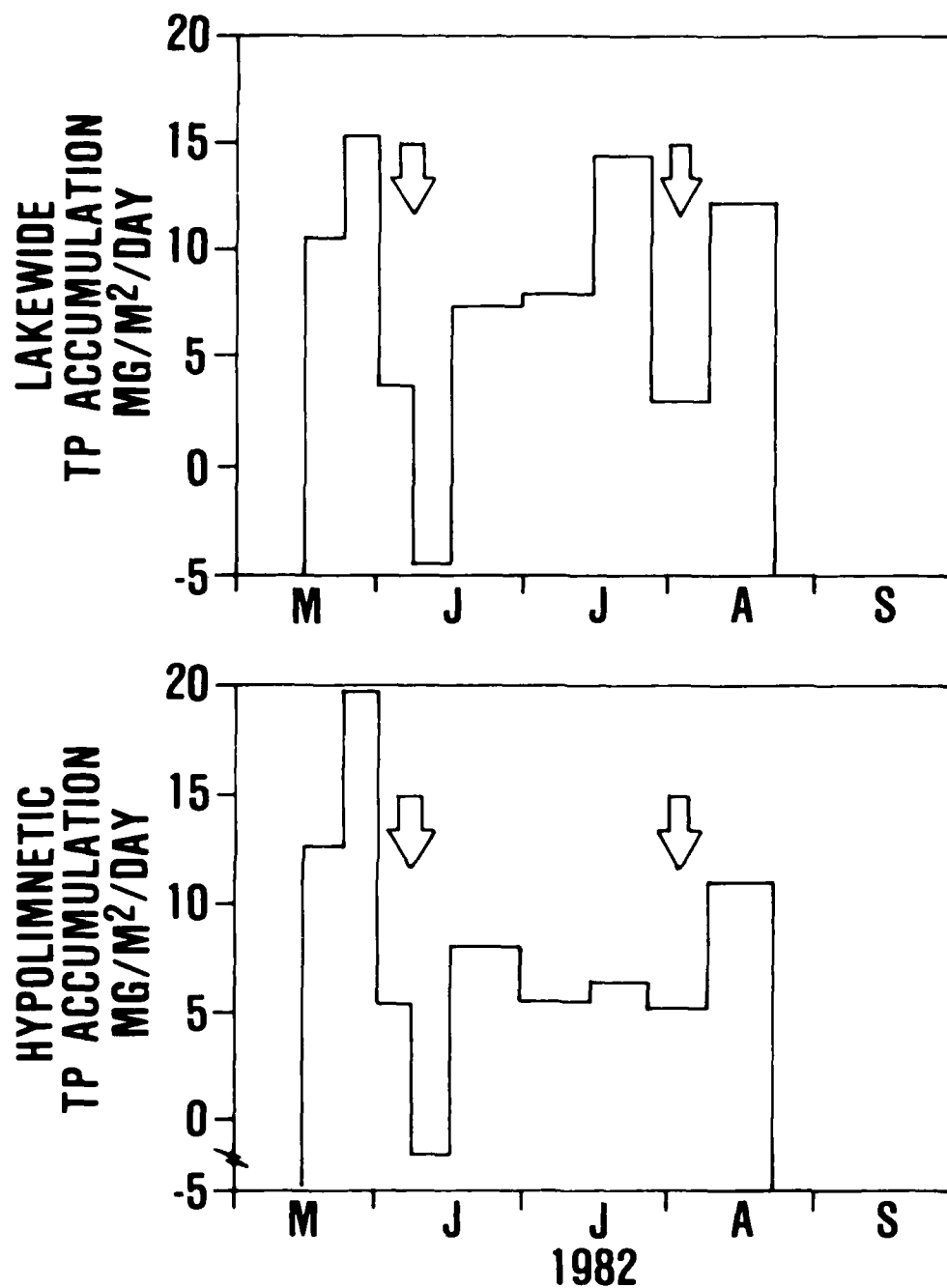


Figure 50. Hypolimnetic and lakewide TP accumulation rates (milligrams/square meter/day)

218. The extent of phosphorus accumulation appeared to be confined to the region near the lower metalimnetic boundary, as suggested by results from close-interval sampling. SRP concentrations decreased markedly above this zone, particularly from June to mid-July. Since the metalimnetic zone is often expansive, upward transport of phosphorus through the metalimnion by diffusion may be a slow process or negligible in Eau Galle Lake. Rather, it appears that the metalimnion acts as a barrier to the extent of phosphorus accumulation, particularly when the lower metalimnetic boundary is stable in the water column (i.e., June-July 1982).

219. The passage of cold fronts and high wind activity caused a pronounced descent of the metalimnion and an introduction of dissolved oxygen to the upper hypolimnion in June and early-August 1982. During these periods, phosphorus was not entrained into the epilimnion, but rather, there was an increase in the deposition of phosphorus and iron in the hypolimnion, and hypolimnetic phosphorus accumulation rates were low. These observations suggested that a relationship between iron, phosphorus, and anoxic conditions possibly played an important role in the fate of phosphorus during large-scale mixing events. It is hypothesized from these results that sufficient quantities of dissolved oxygen were entrained into the upper hypolimnion to cause coprecipitation and deposition of oxidized iron and phosphorus from the water column. Furthermore, a decrease in hypolimnetic volume, as a result of thermocline depression, and the loss of some hypolimnetic phosphorus resulted in a decrease in hypolimnetic phosphorus accumulation rates.

220. The significance of a disruption of hypolimnetic phosphorus dynamics is that internal phosphorus loading to the epilimnion may be temporarily reduced. In Eau Galle Lake, Gaugush (1986) has recently shown that vertical phosphorus transport is an important phosphorus source to the epilimnion during the summer. He observed that marked increases in epilimnetic phosphorus occurred by internal loading when wind activity and mixing caused only a slight depression of the thermocline and did not disrupt anoxic conditions in the hypolimnion appreciably. Large-scale mixing events can reduce this epilimnetic income, when

dissolved oxygen is entrained into the upper hypolimnion and phosphorus is removed from the water column by deposition.

221. The hypolimnetic withdrawal structure also played an important role in hypolimnetic phosphorus dynamics. The output of phosphorus accounted for a large percentage of the hypolimnetic internal load in summer. Vertical SRP concentrations decreased markedly above the withdrawal structure, suggesting that hypolimnetic phosphorus accumulation above this zone was strongly affected by the output of phosphorus. Since the sediment is an important internal source of phosphorus in Eau Galle Lake, hypolimnetic phosphorus accumulation and vertical transport to the epilimnion might have been more substantial in the absence of a hypolimnetic withdrawal.

References

American Public Health Association. 1980. Standard methods for the examination of water and wastewater, 15th ed. Washington, DC. 1,193 pp.

Barko, J. W., D. J. Bates, G. J. Filbin, S. M. Hennington, and D. G. McFarland. 1986. Seasonal growth and community composition of phytoplankton. In R. H. Kennedy and R. C. Gunkel, Jr., eds. Limnological studies at Eau Galle Lake, Wisconsin; Report 2, Special studies and summary. Technical Report E-85-2. US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Blakar, I. A. 1979. A close-interval sampler with minimal disturbance properties. Limnology and Oceanography 24:983-988.

Gaugush, R. F. 1986. Mixing events in Eau Galle Lake. In R. H. Kennedy and R. C. Gunkel, Jr., eds. Limnological studies at Eau Galle Lake, Wisconsin; Report 2, Special studies and summary. Technical Report E-85-2. US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Imboden, D. M., and S. Emerson. 1978. Natural radon and phosphorus as limnologic tracers: horizontal and vertical diffusion in Greifensee. Limnology and Oceanography 23:77-89.

James, W. F. 1986. Seasonal patterns of sediment deposition in Eau Galle Lake. In R. H. Kennedy and R. C. Gunkel, Jr., eds. Limnological studies at Eau Galle Lake, Wisconsin; Report 2, Special studies and summary. Technical Report E-5-2. US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Kortmann, R. W., D. D. Henry, A. Kuether, and S. Kaufman. 1982. Epilimnetic nutrient loading by metalimnetic erosion and resultant algal responses in Lake Waramaug, Connecticut. *Hydrobiologia* 92:501-510.

Larson, D. P., D. W. Schultz, and K. Malueg. 1981. Summer internal phosphorus supplies in Shagawa Lake, Minnesota. *Limnology and Oceanography* 26:740-753.

Mortimer, C. H. 1971. Chemical exchanges between sediments and water in the Great Lakes - speculations on probable regulatory mechanisms. *Limnology and Oceanography* 16:387-405.

Stauffer, R. E., and D. E. Armstrong. 1984. Lake mixing and its relationship to epilimnetic phosphorus in Shagawa Lake, Minnesota. *Canadian Journal of Fisheries and Aquatic Sciences* 41:57-69.

Stauffer, R. E., and G. F. Lee. 1973. Role of thermocline migration in regulating algal blooms. Pages 73-82 in E. J. Middlebrooks, D. H. Falkenburg, and T. E. Maloney, eds. *Modeling the Eutrophication Process*. Ann Arbor Science Publishers, Ann Arbor, Mich.

Wodka, M. C., et al. 1983. Diffusivity-based flux of phosphorus in Onondaga Lake. *ASCE Journal of Environmental Engineering* 109:1403-1415.

PART X: MIXING EVENTS IN EAU GALLE LAKE*

Introduction

222. Weather-related mixing events have been shown to act as an important mechanism for epilimnetic internal nutrient loading in lakes during the summer when external loadings can be expected to be minimal. Stauffer and Lee (1973) demonstrated that cold front passage and wind stress resulted in thermocline migrations in Lake Mendota. These migrations acted to increase epilimnetic nutrient concentrations and were followed by increased chlorophyll concentrations. Stefan and Hanson (1981) observed significant phosphorus transport from the anoxic hypolimnion to the epilimnion associated with mixing in five shallow lakes in south-central Minnesota. Phosphorus transport was followed by intense algal blooms in these lakes. Kortmann et al. (1982) reported the occurrence of algal blooms in response to the thermocline descending below the anaerobic interface in Lake Waramaug.

223. The influence of weather-induced mixing events in reservoirs has not been considered. A comparison of 309 natural lakes and 107 US Army Corps of Engineers (USAE) reservoirs included in the 1972-1975 US Environmental Protection Agency National Eutrophication Survey indicated that reservoirs are generally larger, deeper, morphologically more complex, and have shorter hydraulic residence times than natural lakes (Thornton et al. 1982). These differences, coupled with the importance of advective and unidirectional transport in reservoirs (Baxter 1977) and the presence of either selective or bottom withdrawal, may act to alter a reservoir's thermal regime in such a way as to make it more susceptible to mixing events.

224. The primary objective of this paper is to demonstrate the occurrence and extent of mixing events and their effect on epilimnetic nutrient load and phytoplankton abundance in Eau Galle Lake. A secondary objective is to determine if the design and operation of this

* Part X was written by Robert F. Gaugush.

reservoir influences its susceptibility to mixing events. Eau Galle Lake is well suited for this determination because it is an atypical USAE reservoir in that it is morphologically similar to natural lakes but does have a low-level gated release which may act to alter its thermal regime.

Methods

225. Routine sampling for nutrients, metals, and chlorophyll a was conducted on a biweekly basis throughout the study period (1981 and 1982). Water samples were collected from the surface to the bottom at 1-m intervals using a pump and hose sampler. In situ measurements of dissolved oxygen and temperature were performed weekly with a Hydrolab surveyor (Hydrolab Corp, Austin, Tex.). Details concerning sampling techniques and the specific methods used for the determination of phosphorus, nitrogen, chlorophyll a, iron, and manganese can be found in Johnson and Lauer (1985). Six stations were sampled on each visit, but only the data derived from the deepest station will be presented here.

226. Meteorologic data were obtained from the nearest first-order weather station located at Minneapolis-St. Paul (National Weather Service). Discharge records for Eau Galle Lake were obtained from the US Geologic Survey.

227. Thermal stability per unit lake surface area is the amount of work required to mix the entire volume of the lake to a uniform temperature (Birge 1915). Stability (S , g-cm/cm²) was calculated from the integral given by Hutchinson (1957)

$$S = A_0^{-1} \int_0^{z_m} - [(z - z_g) A_z (1 - \rho_z)] dz$$

where

A_0 = lake surface area, m^2

z_m = maximum depth

z = depth, m

A_z = area enclosed by contour of depth z

ρ_z = density of water at depth z

and the center of gravity (z_g) of the lake is

$$z_g = V^{-1} \int_0^{z_m} z A_z dz$$

where V = lake volume, m^3 .

Volume-weighted mean lake temperature is defined as

$$T_L = V^{-1} \int_0^{z_m} T_z V_z dz$$

where

T_z = temperature at depth z

V_z = stratum volume at depth z

228. Lake heat content, the store of heat that it could impart to its surroundings on cooling to $0^\circ C$, is defined as

$$H_L = c T_L V$$

where c = specific heat of water, $10^3 \text{ kcal/}^\circ\text{C/m}^3$.

229. Relative thermal resistance to mixing (RTRM) (Vallentyne 1957) was obtained by converting water temperature to density and calculating the density difference between two adjacent strata relative to the density difference between water at 5° and 4° C :

$$\text{RTRM}_z = (\rho_z - \rho_{z+1}) / (\rho_{5^\circ \text{ C}} - \rho_{4^\circ \text{ C}})$$

Results and Discussion

230. The development of Eau Galle Lake's heat content generally follows the typical pattern for north temperate lakes (Figure 51). Heat content is at a minimum when the lake is ice-covered and begins to increase with ice-out, reaching a maximum sometime in July. With the onset of cooler weather, the lake begins to lose heat and reaches minimum heat content in early December. Development of stability (Figure 51) was highly correlated with the changes in heat content ($r = 0.086$, $p < 0.001$). Eau Galle Lake begins to stabilize in mid-April, attains maximum stability at or a few weeks prior to maximum heat content, and reaches a minimum in mid-October.

231. The months of June, July, and August of both 1981 and 1982 were marked by a series of decreases in stability, and the large destabilizations are associated with considerable heat loss. Significant heat losses and decreased stability might be expected in late-August, but the months of June and July should be characterized by a steady increase in heat content and stability.

232. Eau Galle Lake has two features which may make it more susceptible to these summer destabilizations. First, its hypsographic, or depth-area, curve (Figure 52) illustrates that the majority of the lake's area (and volume) lies above the 4-m contour. The predominance of depths less than 4 m results in a very shallow center of gravity ($z_g = 2.7 \text{ m}$). Given otherwise equal conditions, stability will increase with the descent of the thermocline, reaching a maximum value when the

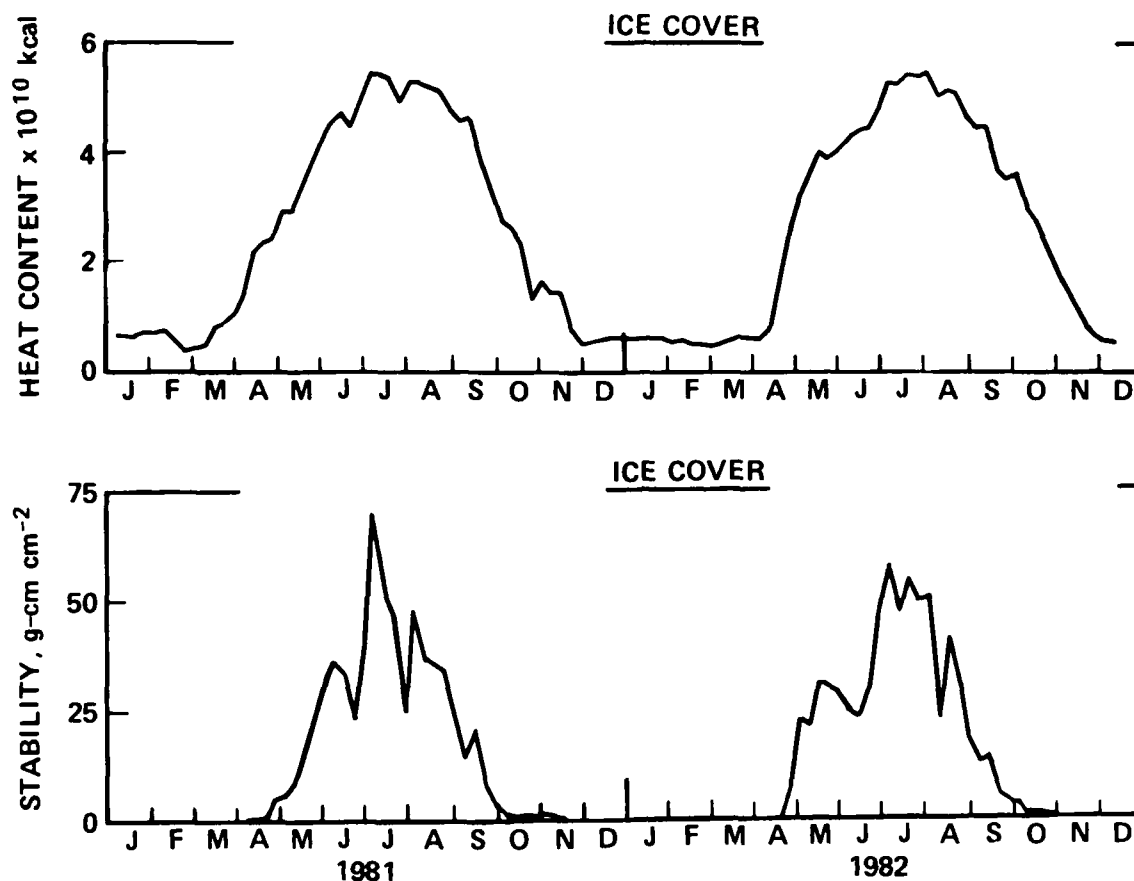
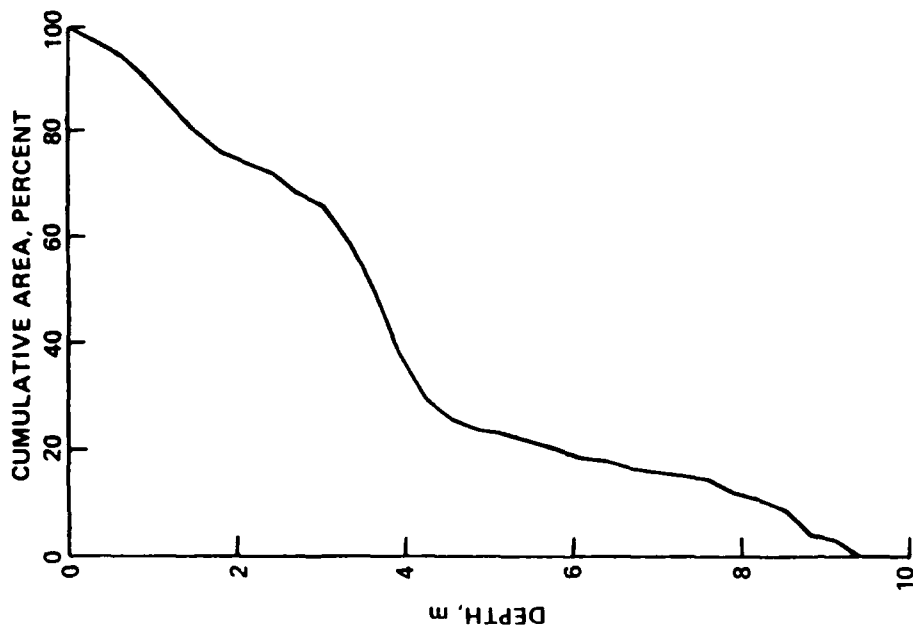
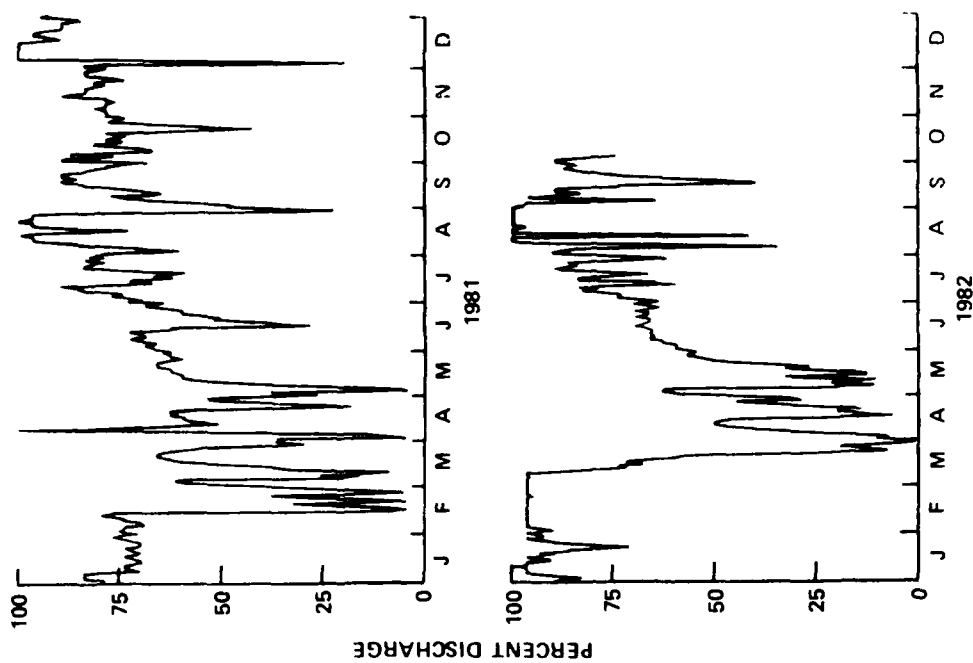


Figure 51. Heat content (upper) and thermal stability (lower) of Eau Galle Lake, 1981 and 1982

thermocline lies at the center of gravity. As the thermocline descends past this depth, stability decreases (Rutner 1963). The shallow center of gravity may make Eau Galle Lake more susceptible to mixing. Second, a considerable fraction of the discharge from Eau Galle Lake is through the low-level release gate located 5 m below the surface (Figure 52). Low-level releases are essentially constant at approximately $3 \times 10^4 \text{ m}^3/\text{day}$, but outflow from the surface changes with pool elevation. Snowmelt and spring runoff raise pool elevation and reduce the relative contribution of the low-level release. During the stratified period (late April through September), the low-level release accounts for 50 percent or more of the discharge. The output of relatively colder water from below the surface can result in considerable hypolimnetic heating as



a. Hypsograph (depth-area curve)



b. Percent discharge

Figure 52. Relative hypsograph for Eau Claire Lake and percent of total discharge due to low-level release

outflowing water is replaced with warmer water from above. Hypolimnetic heating is evident from the progression of isotherms below 6 m (Figure 53) and acts to reduce the density differences from surface to bottom, which in turn reduces stability.

233. The combination of a shallow center of gravity and considerable hypolimnetic heating make Eau Galle Lake particularly susceptible to destabilization during its stratified period. Figure 54 presents the meteorologic conditions associated with changes in stability and mean lake temperature. Meteorologic variables have been smoothed using a 3-day moving average. Periods of destabilization were generally accompanied by decreases in both maximum and minimum air temperature, increasing barometric pressure, winds out of the northwest, and precipitation. West-central Wisconsin is affected by disturbances originating in the northwest, which migrate eastward and are followed by colder polar air masses (National Weather Service). As a cold front passes there is an increase in barometric pressure, an abrupt drop in air temperature, an increase in wind velocity, and shift from southwest winds to winds out of the northwest (Blair and Fite 1957). Surface heat loss during the clear, cool nights that follow the passage of a cold front will result in convection currents that can act to a depth of 3 m (Wetzel 1975). Once convection currents have decreased stability, winds can act to mix a considerable fraction of the lake's volume. In Eau Galle Lake, decreases in stability occur in one of two ways: (a) with a large decrease in T_L (significant heat loss), or (b) with little or no change in T_L (a redistribution of heat). It will be shown that these different types of destabilization have profoundly different effects on epilimnetic nutrient concentrations and phytoplankton abundance.

234. Decreases in stability are indicative of mixing, but stability, as defined, expresses the condition of the lake as a whole and cannot be used to determine the extent of a mixing event. Relative thermal resistance to mixing, which is calculated for each depth stratum, can indicate the depth to which mixing has occurred. A temperature change of 1 °C/m in the range of 10° to 30° C produces RTRM values ranging from 10 to 37. An RTRM value of 30 was found to most closely

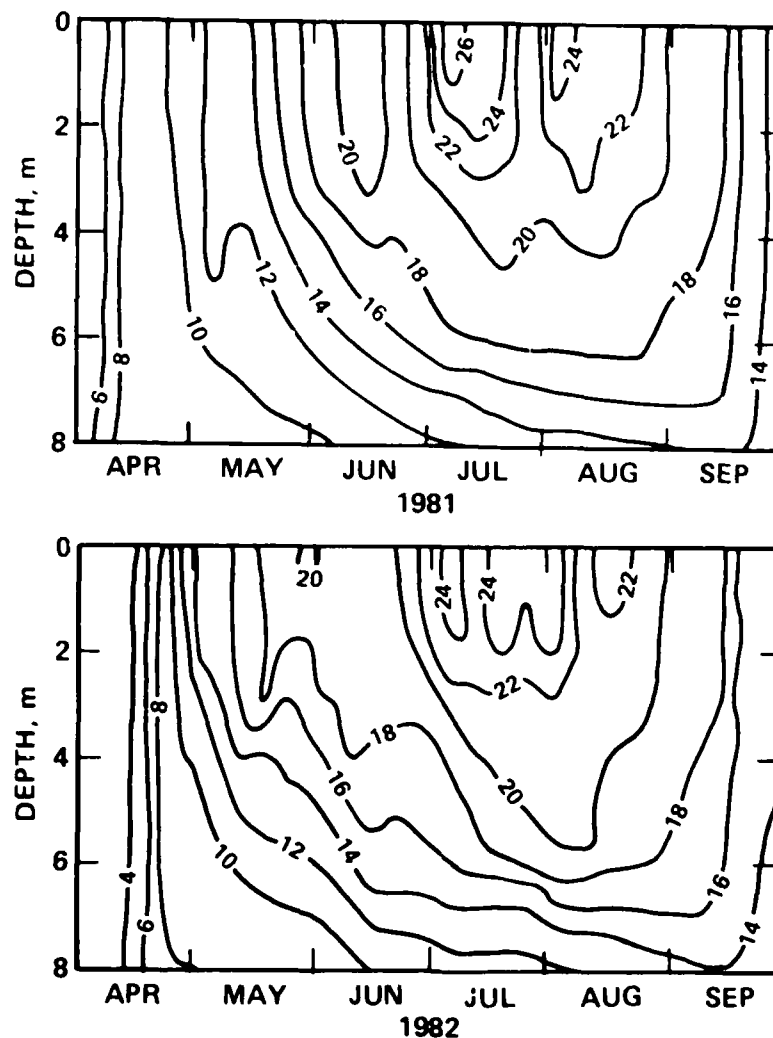


Figure 53. Isotherms ($^{\circ}\text{C}$) for April through September in 1981 (upper) and 1982 (lower)

approximate the position of the thermocline (Kortmann et al. 1982), and that convention was adopted for Eau Galle Lake.

235. Thermocline depth ($\text{RTRM} = 30$) was extremely dynamic in Eau Galle Lake during its stratified period (Figure 55), and the extent of hypolimnetic anoxia (Figure 55) closely followed changes in thermocline depth. Hypolimnetic anoxia increased during periods when the thermocline was high in the water column. Large-scale mixing events, characterized by a large, rapid descent of the thermocline, introduced oxygen into the hypolimnion and reduced the extent of anoxic conditions.

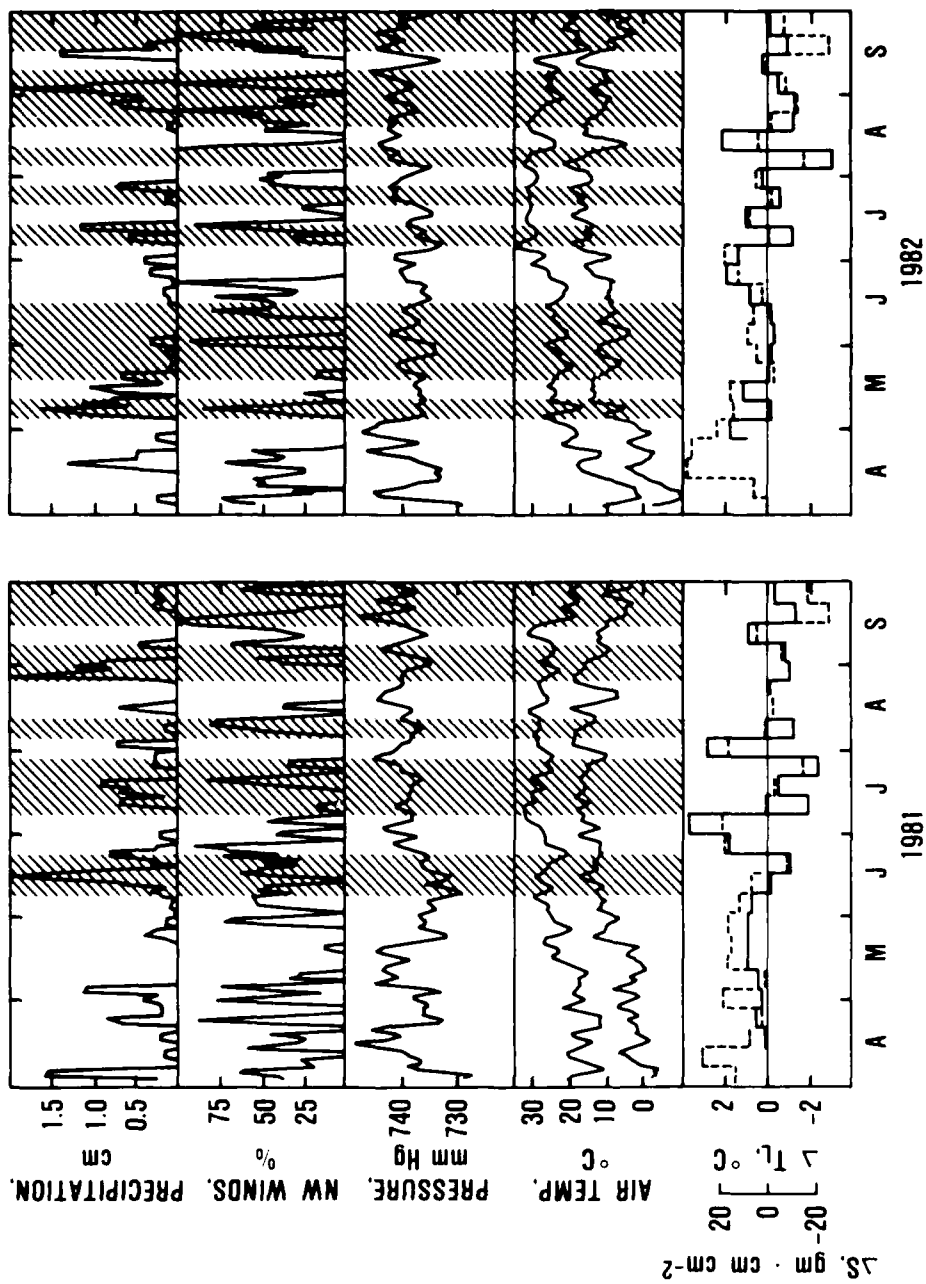


Figure 54. Meteorologic conditions associated with changes in stability (ΔS , —) and mean lake temperature (ΔT_L , ---) for April through September 1981 (left) and 1982 (right). Shaded areas correspond to periods of decreasing stability

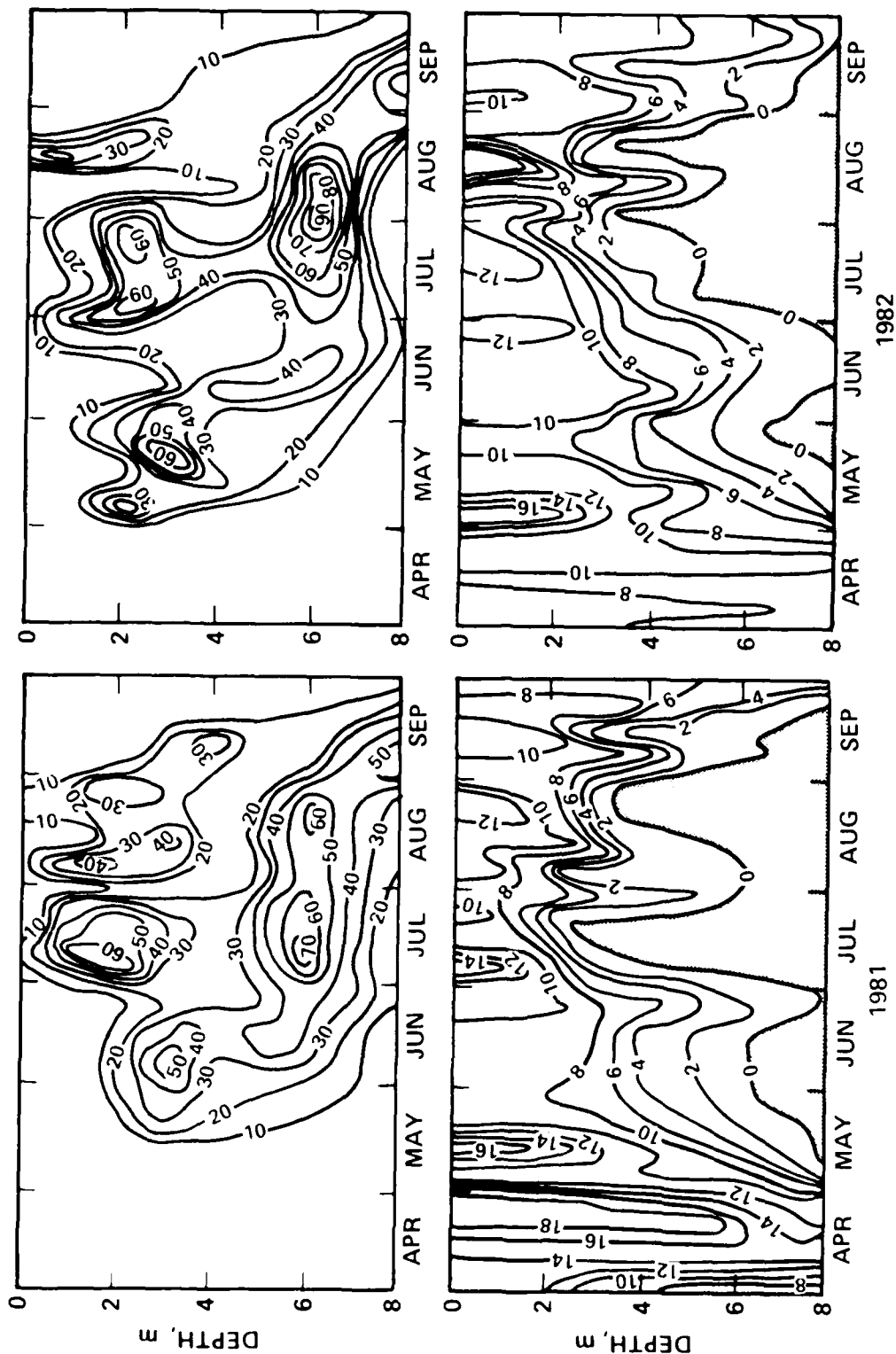


Figure 55. Isopleths of RTRM (upper) and dissolved oxygen (mg/L, lower) for April through September 1981 (left) and 1982 (right)

Note that when anoxia is at a maximum (peaks in July and August), the epilimnion is separated from the hypolimnion by no more than 1 m. Small-scale mixing events during maximum anoxia exhibited only a slight descent of the thermocline and had no effect on the extent of anoxia, but can be expected to easily transport nutrients from the hypolimnion and result in an increase in phytoplankton abundance.

236. Phytoplankton dynamics (expressed by changes in chlorophyll a concentrations) are clearly influenced by changes in thermocline depth and its proximity to the anoxic hypolimnion (Figure 56). Both years exhibited a spring bloom prior to the onset of stratification and hypolimnetic anoxia. Later blooms occurred when the extent of anoxia was at a maximum and the nutrient-rich hypolimnion lay relatively close to the surface. These blooms were separated by large-scale mixing events which acted to reduce surface concentrations of chlorophyll a. Decreases in chlorophyll a concentrations in response to a large-scale mix were the result of two factors: (a) an initial dilution of epilimnetic concentrations due to the expansion of the epilimnion, and (b) if mixing continues and the mixed layer extends below the photic zone, the phytoplankton will spend a greater amount of time under poor light conditions, thereby reducing growth rates (Stefan, Skoglund, and Megard 1976).

237. The stratified period for Eau Galle Lake can be separated into a series of three different events based on changes in stability, heat content, and the extent of anoxia. A stable period can be identified by increases in stability, heat content, and anoxia. Mixing periods are characterized by losses in stability and heat content. Large- and small-scale mixes can be differentiated based on the large loss of heat and the decrease in the extent of anoxia associated with large-scale mixes. Representative changes associated with these three events are presented in Table 15.

238. Stable periods were typified by surface losses of both total phosphorus and nitrogen while greater depths exhibit considerable gains. Epilimnetic losses were primarily the result of particulate matter settling out during these relatively calm periods. Concentrations of

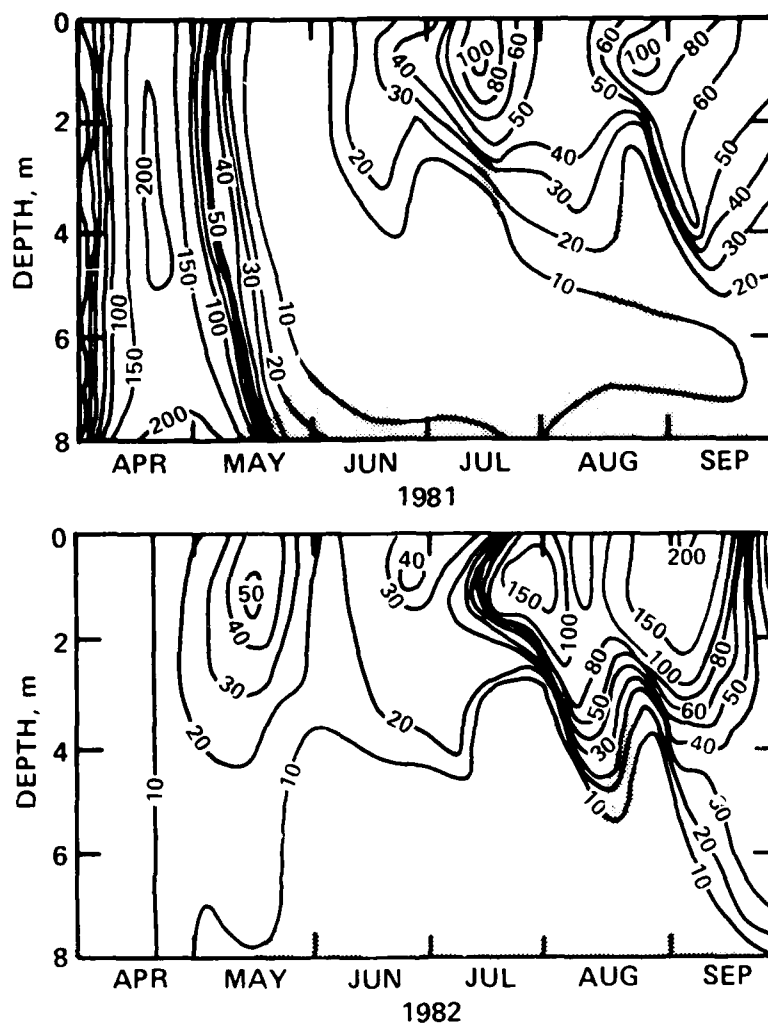


Figure 56. Isopleths of chlorophyll a concentrations (mg/m^3), April through September 1981 (upper) and 1982 (lower)

dissolved metals generally showed gains as would be expected given the increased extent of hypolimnetic anoxia. Surface concentrations of chlorophyll a either showed no change or decreased during stable periods.

239. Significant epilimnetic loading of nitrogen and phosphorus occurred during small-scale mixes. Internal loading rates during these mixes were one to two orders of magnitude higher than average external loading rates. External loading during the late spring and summer

Table 15

Changes Associated with Stable Periods and Large- and Small-Scale Mixing Events*

Year	Stable		Mixing Events			
	1981		Small-Scale		Large-Scale	
	1981	1982	1981	1982	1981	1982
Time period (month/day)	5/19 - 6/2	6/15 - 6/29	6/30 - 7/14	7/13 - 7/27	7/14 - 7/28	7/27 - 8/10
Δ Stability (g-cm/cm ²)	14.5	23.0	-18.0	- 5.7	-22.1	- 27.4
Δ Heat content ($\times 10^9$ kcal)	4.3	3.3	- 0.1	- 0.1	- 4.5	- 4.4
Δ Anoxic volume ($\times 10^4$ m ³)	1.1	1.4	0.0	0.0	-10.4	- 5.3
Total phosphorus						
0-3 m	-0.2	- 1.0	10.3	8.9	- 0.2	- 2.6
4-8 m	3.5	33.5	11.3	3.5	- 6.3	0.2
Total nitrogen						
0-3 m	-6.0	-40.8	37.7	267.8	-13.2	-177.3
4-8 m	4.7	19.8	24.1	74.8	- 8.7	- 35.6
Dissolved iron**	0.0	17.5	50.3	24.2	-33.0	9.3
Dissolved manganese**	17.9	2.3	5.6	10.6	-22.3	- 10.1
Chlorophyll <u>a</u> (mg/m ³ /day)†	0.0	- 1.3	7.6	5.6	- 4.0	-0.8

* Units are milligrams per square meter unless otherwise noted.

** Calculations based on entire lake (0-8 m).

† Calculations based on 0-3 m.

averaged about $0.02 \text{ mg P/m}^2/\text{day}$ and $0.2 \text{ mg N/m}^2/\text{day}$.^{*} Net positive internal loading of nutrients to the epilimnion resulted in increased chlorophyll a concentrations during small-scale mixes. These mixes had little effect on the hypolimnion given that concentrations of nitrogen, phosphorus, and dissolved metals exhibited gains during these events.

240. Large-scale mixes acted as short-lived turnovers and resulted in epilimnetic losses of phosphorus, nitrogen, and chlorophyll. The effects of the introduction of oxygen associated with these mixes were evident in the general loss of phosphorus, nitrogen, and dissolved metals from the hypolimnion. Although phosphorus and dissolved iron showed positive changes during a large-scale mix in 1982, the rate of change is considerably lower than rates observed during small-scale mixes.

Conclusions

241. Two different types of mixing events can be identified in Eau Galle Lake. First, there are small-scale mixes that occur when the epilimnion and the anoxic, nutrient-rich hypolimnion lie relatively close together. The effects of these mixes are similar to those reported for natural lakes, with an internal loading of nutrients to the epilimnion and a resultant increase in phytoplankton abundance. Second, large-scale mixes occur that function essentially as short-lived turnovers. With the introduction of oxygen into the previously anoxic hypolimnion, nutrient and dissolved metal concentrations decrease in the lake as a whole. These large-scale mixes have not been reported for natural lakes and may be a result of lower stability in Eau Galle Lake due to the effect of low-level releases on its thermal regime.

^{*} Personal Communication, 1986, R. H. Montgomery, Department of Civil Engineering, Colorado State University, Fort Collins, Colo.

References

- Baxter, R. M. 1977. Environmental effects of dams and impoundments. *Ann. Rev. Ecol. Syst.* 8:255-283.
- Birge, E. A. 1915. The heat budgets of American and European Lakes. *Trans. Wis. Acad. Sci. Arts Lett.* 18:166-213.
- Blair, T., and R. Fite. 1957. *Weather elements*. Prentice-Hall, Inc., Englewood Cliffs, N. J. 414 pp.
- Hutchinson, G. E. 1957. *A treatise on limnology; Vol I. Geography, physics and chemistry*. J. Wiley, Inc., New York. 1,015 pp.
- Johnson, D., and Lauer. 1985. General methods. In R. H. Kennedy, ed. *Limnological studies at Eau Galle Lake, Wisconsin; Report 1, Introduction and water quality monitoring studies*. Technical Report E-85-2. US Army Engineer Waterways Experiment Station, Vicksburg, Miss.
- Kortmann, R. W., D. D. Henry, A. Kuether, and S. Kaufman. 1982. Epilimnetic nutrient loading by metalimnetic erosion and resultant algal responses in Lake Waramaug, Connecticut. *Hydrobiologia* 92:501-510.
- Rutner, F. 1963. *Fundamentals of limnology*. University of Toronto Press, Toronto. 295 pp.
- Stauffer, R. E., and G. F. Lee. 1973. The role of thermocline migration in regulating algal blooms. Pages 73-82 in E. J. Middlebrooks, D. H. Falkenberg, and R. E. Maloney, eds. *Modeling the eutrophication process*. Ann Arbor Science Publishers, Ann Arbor, Mich.
- Stefan, H., and M. J. Hanson. 1981. Phosphorus recycling in five shallow lakes. *ASCE Journal of the Environmental Engineering Division* 107:713-730.
- Stefan, H., T. Skoglund, and R. O. Megard. 1976. Wind control of algae growth in eutrophic lakes. *ASCE Journal of the Environmental Engineering Division* 102:1201-1213.
- Thornton, K. W., R. H. Kennedy, A. D. Magoun, and G. E. Saul. 1982. Reservoir water quality sampling design. *Water Resources Bulletin* 18:471-480.
- Vallentyne, J. R. 1957. *Principles of modern limnology*. American Scientist 45:218-244.
- Wetzel, R. G. 1975. *Limnology*. W. B. Saunders Co., Philadelphia, Pa. 743 pp.

PART XI: SUMMARY*

242. Water quality studies were conducted at Eau Galle Lake during the period October 1980 to September 1982 as part of Task VIIA of the Environmental and Water Quality Operational Studies Program. The overall purpose of these EWQOS studies was to gain a better understanding of relations between project design, construction and operation, and water quality. Specific objectives for studies at Eau Galle Lake included: the establishment of a water quality data base for the calibration and verification of a mathematical model; evaluation of nutrient and metal dynamics; delineation of relations between material inputs, sedimentation, recycling, and losses by discharge; identification of factors affecting primary and secondary production; and evaluation of physical factors influencing lake dynamics.

243. Eau Galle Lake is a small, shallow, flood control reservoir that receives high and seasonally variable nutrient loads and exhibits symptoms of eutrophication, including hypolimnetic anoxia, abundant growths of rooted aquatic macrophytes, excessive algal production, and reduced water clarity.

244. Dissolved oxygen concentrations decline in bottom waters immediately following the onset of thermal stratification (mid- to late-May), and anoxic conditions are established by early to mid-June; by July, anoxic conditions are observed at depths below 3 to 4 m. Autumnal circulation and reoxygenation of bottom waters begins in mid- to late-September and is complete by early October.

245. A submersed aquatic macrophyte community is well established in the lake with highest plant densities occurring in shallow areas adjacent to the point of inflow of the Eau Galle River and in littoral areas of a cove located along the western shore. Dominant species are *Ceratophyllum demersum* L. and *Potamogeton pectinatus* L. Peak standing crop occurs in mid-August, and two periods of senescence are observed--one in early spring following ice-out and one in fall. This community

* Part XI was written by Robert H. Kennedy and Robert C. Gunkel, Jr.

provides a rich habitat for invertebrates and fish and is a potential source of nutrients for open-water areas of the lake. Based on nutrient concentrations of plant tissues and areal coverage measurements, it is estimated that plants may represent a pool of as much as 400 to 450 kg nitrogen and 60 to 70 kg phosphorus during periods of peak standing crop. While a portion of this pool would be released following senescence and decay, comparisons of the above quantities with estimates of external and other internal loading sources suggest that macrophytes have only a minor impact on the nutrient budget of this lake.

246. Chlorophyll a concentrations, an approximate measure of phytoplankton abundance, are seasonally high (maxima of 100 to 150 mg/m³) and exhibit two annual peaks. The first, attributable to diatoms, occurs in the spring following ice-out but preceding the establishment of stable thermal stratification. The timing and magnitude of the peak are influenced to a great extent by the date of ice loss and by the occurrence and magnitude of snowmelt and runoff. Peak concentrations following ice-out in 1982, a year in which spring runoff and associated material loading was rapid and excessive, were considerably lower (25 versus 150 mg/m³) than for the same period in 1981.

247. Chlorophyll a concentrations during the summer stratified period are typical for eutrophic lakes and are strongly influenced by physical events. The growth of diatoms, which were dominant during the spring of 1981 and 1982 when the lake was unstratified and the water column well mixed, was markedly reduced by warmer temperatures and increased stability of the water column. Dominant during the summer months were species of blue-green algae, which often formed blooms and surface scums.

248. The growth of algal populations during the summer stratified period were regulated, in part, by the occurrence of mixing events. Eau Galle Lake is susceptible to periodic mixing due to the release of cool, hypolimnetic waters and the resulting loss in thermal gradient. Because of the weak thermal gradient, wind shear associated with meteorologic events is often sufficient to cause exchanges of water and materials in the vicinity of the thermocline. While these events are not sufficient

to bring about complete mixing of the water column, significant quantities of growth-stimulating nutrients are made available to algal populations in surface layers. Since nutrient inputs to the lake from external sources are minimal during the summer, low-flow months, these events are critical in maintaining a large algal standing crop.

249. The transport of nutrients from the hypolimnion to the epilimnion is made possible by the release of these materials from bottom sediments and their accumulation in the hypolimnion. Marked gradients in the concentrations of phosphorus and nitrogen were documented during the summers of 1981 and 1982. Concentrations were highest in water layers immediately above the sediment/water interface due to solubilization and upward diffusion. While declining in layers more distant from the sediment/water interface, concentrations near the thermocline were well above those observed above the thermocline.

250. External loading, which ultimately dictates the quantity of material present in the lake, displayed marked seasonal variability. Loads and concentrations were highest during and immediately following the spring snowmelt and runoff period, and lowest during the summer months. The Eau Galle River, which drains a predominantly forested and pastured watershed, provided a majority of the annual water and material inputs to the lake.

251. While the large annual loads of phosphorus and nitrogen are consistent with the current trophic state of the lake and are similar to those for other lakes in the area, seasonality plays a major role in the development of eutrophic conditions in this lake. High external loads, which enter the lake in early spring, and physical conditions in the lake prompt the rapid growth of diatoms in spring. The sedimentation of nutrient-containing particulates carried to the lake by tributaries and senescent algal cells results in the storage of large quantities of readily available nutrients in bottom sediments. As thermal stability is established, diatom populations decline, anaerobic conditions follow the decay of organic materials, and materials such as phosphorus, iron, and manganese are reintroduced to the water column. Subsequent periodic mixing transports these materials to surface waters during periods when

external nutrient loads are minimal. The result is the occurrence of nuisance algal conditions.

END

7-87

DTIC